At Risk

VOLUME ONE

Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk

University College of the Cariboo
Kamloops, BC, February, 15-19, 1999

Edited by Laura M. Darling

British Columbia Ministry of Environment, Lands & [illegible]
University College of the Cariboo

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101-03808/

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February 15 - 19, 1999
University College of the Cariboo
Kamloops, British Columbia

edited by
Laura M. Darling
Wildlife Branch
British Columbia Ministry of Environment, Lands and Parks
Victoria, British Columbia

August 2000

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Canadian Cataloguing in Publication Data
Main entry under title:
At risk: proceedings of a conference on the biology and management of species and habitats at risk.
Includes bibliographical references.
ISBN 0-7726-4348-2

1. Endangered species - British Columbia - Congresses.
2. Biological diversity conservation - British Columbia - Congresses.
3. Wildlife habitat improvement - British Columbia - Congresses.
I. Darling, Laura M. II. University College of the Cariboo. III. British Columbia. Ministry of Environment, Lands & Parks.
QH77.C3A8 2000 333.95'22'09711 C00-960290-9

Cover artwork by Michael Hames

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PREFACE

This collection of papers stems from the third conference in British Columbia to focus on species and habitats at risk. The first, co-hosted by the Federation of British Columbia Naturalists, Douglas College, and the British Columbia Ministry of Environment was held at Douglas College in Vancouver in 1980 (Stace-Smith et al. [eds.], 1980, *Threatened and Endangered Species and Habitats in British Columbia and the Yukon*). The second was sponsored by the Federation of British Columbia Naturalists and the Northwest Wildlife Preservation Society and was held in Vancouver in 1991 (Rautio [ed.], 1991, *Community Action for Endangered Species*). This most recent meeting, held in Kamloops, 15 – 19 February 1999, and co-hosted by the British Columbia Ministry of Environment, Lands and Parks and the University College of the Cariboo, had more than 60 partnering organizations and businesses. Interest and participation in each of these conferences was high and attendance was at capacity. The papers in the proceedings from these 3 conferences can be used to gauge of the state of our knowledge of the biology of species. At each of the first 2 conferences, 8 papers dealt with the biology of a species at risk. In this third set of proceedings, there are papers dealing with aspects of the biology of more than 60 species. Some of the titles of the papers presented in Kamloops would have been unimaginable at the earlier conferences—for example, an overview of research on the marbled murrelet would have been a very slim paper a decade or 2 ago!

The proceedings from the Kamloops conference also hint at other emerging trends in approaches to species and habitats at risk. In the past decade, increasing concern about loss of biodiversity and the growing number of species at risk has resulted in an increase in research and inventory work, especially for species groups and habitats that have been traditionally ignored by wildlife management. As a result, included in these proceedings are preliminary lists of species at risk for mosses, marine invertebrates, dragonflies, beetles, freshwater molluscs, and butterflies. "Natural history" has also resurfaced as an important field of study as such information becomes crucial to the proper management of species, particularly those at risk, and this renaissance is reflected in a number of papers. An expanding awareness of the role of organisms at risk in ecological processes is highlighted in several places in these proceedings. Examples include discussions of the fragile cryptogamic crusts of grasslands and shrub-steppes, and of the complicated and fragile interconnections of mycorrhizal fungi.

The Kamloops conference and its proceedings are an important step in communicating the results of the recent, original and innovative work to managers, scientists, educators, naturalists, and the general public. There are many sobering signs of how desperate the situation has become: far more extensive and complicated endangered species lists; a growing number of ecosystems at risk; an increasing understanding of how fragile some ecosystems are; and accelerated rates of habitat loss and fragmentation. On the bright side, there is the tremendous energy that has in recent years been put toward public education, and documenting, classifying, researching, and conserving species at risk.

These papers are a valuable glimpse into that effort.

David Fraser and Karl Larsen, Co-chairs

Eric Lofroth, Don Reid, Laura Darling, and Eric Parkinson, Organizing Committee

EDITOR'S NOTE

The papers in these Proceedings have not been peer reviewed – the content is fully the authors' responsibility. Each paper was subjected to a grammar and copy edit, to address issues of spelling, grammar, and conventions of style and layout selected for this document. I made it my responsibility to review all edit suggestions made by our copy editors, and to approve all changes I felt did not alter the intent of the author; most authors were contacted during this process to address contentious edits, to clarify wording, or to provide missing documentation. The papers were submitted at varying degrees of technical detail, and individual writing styles varied substantially; however, we made no attempt to change the character of the papers submitted. Graphics were submitted in numerous production formats and layout styles; we have done our best to ensure that each graphic is readable and gets the point of the graphic to the reader. The final product is the result of the immense effort by many people to make each paper better.

LAURA M. DARLING, EDITOR

ACKNOWLEDGEMENTS

Many hands make light work. These Proceedings and the conference from which they stem could not have happened without the input and support of many individuals and organizations.

The conference, sponsored by the B.C. Ministry of Environment, Lands and Parks and the University College of the Cariboo, was supported by financial and in-kind contributions from over 25 agencies and organizations, whose logos are presented on page iii of this volume.

Publication of these Proceedings was supported by financial contributions from 9 agencies and organizations whose logos appear on page ii. Special thanks to BC Hydro for making a substantial cash contribution, and to Ed Hill for making it happen. Authors of lengthy papers were called upon to contribute 'page charges' to cover costs of editing and layout. Additional financial support was provided by John Carnio, Curator of Mammals, The Toronto Zoo. Many thanks to Queen's Printer for rock-bottom pricing for print production, facilitating greater distribution of these documents.

The proceeds of a silent auction held at the conference also were directed towards the publication of these proceedings. The following organizations and individuals generously donated items: The Nature Trust of British Columbia, Nature Conservancy Canada, Ian Goudie, Bear Brewing Company, Lone Pine Press, Northwest Wildlife Preservation Society, Fraser's Thimble Farms, David Fraser, Neville Crosby Inc., Eric Lofroth, Vancouver Aquarium, Fred Cooke, Western Canada Wilderness Committee, Goldstream Park Nature House, Columbia Basin Fish and Wildlife Compensation Program, Greystone Press, Mountain Equipment Coop, UBC Press, Parks Canada, Wild Birds Unlimited, B.C. Conservation Data Centre, Vancouver Telescope Center, Royal B.C. Museum, Habitat Acquisition Trust, Creston Valley Wildlife Management Area, Province of B.C. Wildlife Tree Committee, David Green, The Field Naturalist, Dick Cannings, Rob Butler, Vancouver Island Marmot Foundation, BC Hydro, The Steelhead Society, Laskeek Bay Conservation Society, Robert Busch, Canadian Forest Products, World Wildlife Fund, Liz Hammond-Kaarremaa, Friends of Ecological Reserves, Jim Pojar, International Pacific Halibut Commission, Crown Publications, Tracee Geernaert, Larry Halvorsen, Bill Harper, and John Krebs.

Thanks to those who ensured the conference ran smoothly: J. A. Wightman for fund raising, conference planning, and administrative management of incoming submissions and conference scheduling; T. Rankin of the Forestry Continuing Studies Network for initial organization of the conference; G. Lyons and the UCC Activity Centre staff who orchestrated minute-to-minute activities at the UCC Campus Activity Centre; A. Harcombe and D. Eastman for input to selection of papers; B. Gowans, for her part in the silent auction; Kamloops Naturalist Club for running field trips at the conference; D. Lewis and R. Morris, for help in preparing conference packages; D. Williams, for organizing the poster session at the conference; session chairpersons for keeping speakers on time; T. Dickinson , B. Heise and W. Klenner, for hosting the keynote speaker and public lecturers; 'Bookies' and 'Strictly For The Birds' for acting as ticket agents for the public lectures; Roger Barnsley (President of UCC), John Jules (Kamloops Indian Band), and Cliff Branchflower (Mayor of Kamloops) for supporting the conference and graciously welcoming delegates; and Bear Brewery, for crafting a special run of 'Northern Spotted Ale' just for the conference.

C. Hoodicoff efficiently coordinated the huge volunteer workforce: N. Bertram, D. Bissonette, J. Bleuschke, A. Callahan, R. Canart, S. Cote, M. Grant, D. Haughland, T. Hoodicoff, J. Hromadnik, K. Johnston, C. Kissick, L. Larson, V. Larson, D. Lewis, K. Lowson, M. MacDonald, R. McAllister, R. Morris, C Nesbit, A. Norquay, J. Pedersen, K. Pedersen, D. Pettman, N Prichard, S. Raven, T. Richens, A. Ross, S. Rowsell, P. Ruley, K. Safford, L. Sheperd, J. Sorensen, B. Tanner, C. Tarasoff, A. Tipler, W. Tokarek, K. Turriff, S. Warman, and C. Yakowski.

Thanks to those who contributed to the production of these Proceedings: Ros Penty and Frances Backhouse, editors, and Larry Grainger, Wayne Sampson and Denton Pendergast of Rocket Science Design for patience, persistence and professionalism to improve the quality of each and every paper in this Proceedings; Gail Harcombe for facilitating and trouble-shooting production; UCC for carrying and administering proceeds from the conference; proof readers and other volunteers who ensured the final product was complete, M. Austin, M. Badry, C. Cadrin, S. Cannings, M. Chutter, S. Eaves, S. Flynn, L. Friis, L. Haggstrom, I. Hatter, J. Hobbs, G. Jones, D. Kang, N. Laity, T. Lea, E. Lofroth, Z. O'Neill, K. Paige, J. Quayle, L. Ramsay, L. Stanlake, H. Schwantje, L. Shaw, C. Tolkamp, B. von Sacken, L. Westereng, K. Wipond.

Thanks to Michael Hames for his artistic contribution to the conference and to species at risk; his art now graces posters, T-shirts and these Proceedings.

And a special recognition to each and every one of the presenters and delegates for participating.

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Species at Risk, Conservation Strategies, and Ecological Integrity

G. G. E. Scudder
Department of Zoology and Centre for Biodiversity Research, University of British Columbia
6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada
scudder@zoology.ubc.ca

ABSTRACT

Species at risk are relatively easy to identify. Saving them is another matter. There are few success stories. With increased human populations, and consequent demands for more space and more resources, there will be an ever increasing number of species at risk. Our current conservation strategies are largely ad hoc, unintegrated, and ineffective. They must be replaced with new strategies focused on saving species at risk in the context of functioning ecosystems, if there is to be any measure of success. We need to assess the effectiveness of current protected areas, and land use planning. Our conservation planning needs to be revised and revitalized. This could be done with some clearly stated objectives, using GIS-based heuristic algorithms. It could establish a minimum set of essential conservation areas, based upon the principles of irreplaceability and complementarity. It could document what we have so far achieved, and what needs to be done in the future. It is obvious that species at risk belong and function in ecosystems, and depend upon the continued existence of their essential habitats in these ecosystems. Just setting aside protected areas is not enough. They and the adjacent areas have to be managed, and managed in an ecosystem context, with the appropriate mix of coarse- and fine-filter habitat management initiatives. Ecosystems are not static. They continually undergo dynamic change, moving through states of exploitation, conservation, release, and reorganization. To save species at risk in these ecosystems, we must guarantee that these ecosystems retain integrity, that is, the ability to maintain function in both the short and long term, in the face of constantly changing conditions. We need to understand the natural disturbance regimes, and recognize the scope, scale, and temporal patterns of change. To save species at risk we must manage our protected and other areas within the framework of ecological integrity, and understand that this can only be accomplished in community and regionally acceptable models.

Key words: conservation strategies, ecological integrity.

SPECIES AT RISK

Species at risk are relatively easy to identify. Using a diverse set of criteria, more than 1,575 such species have been listed in British Columbia, although very few have legal protection.

Only 4 species, namely the burrowing owl, sea otter, American white pelican, and Vancouver Island marmot, are legally designated under the British Columbia Wildlife Act (SOE 1998). The peregrine falcon and the bald eagle are listed in Appendix I under CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora; Environment Canada 1997), and several species, including the white sturgeon, sandhill crane, black bear, and grizzly bear are listed in Appendix II. The British Columbia Fish Protection Act, passed in 1997, provides legal protection for some aquatics, but so far no list has been promulgated.

Three extinct (Dawson's caribou or the Queen Charlotte Island population of the woodland caribou, and 2 Hadley Lake sticklebacks), 3 extirpated (sage grouse, pygmy short-horned lizard, and island marble butterfly), 21 endangered, 24 threatened, and 43 vulnerable species are listed for British Columbia by the Committee on the Status of Endangered Wildlife in Canada[1] (COSEWIC; World Wildlife Fund Canada 1998). There are 74 vertebrate taxa on the provincial Red List (candidates for legal designation as endangered or threatened), and 78 vertebrate taxa on the Blue List (species considered vulnerable or sensitive; Cannings 1998). In addition there are 234 Red-listed and 356 Blue-listed vascular plants (S. Cannings, British Columbia Conservation Data Centre, pers. comm.), and more than 800 invertebrates potentially rare and endangered (Scudder 1994, 1996*a*), with some already included on the Red and Blue lists.

The COSEWIC-listed species for the most part do not have any legal status, as Canada has yet to pass endangered species legislation. Similarly, the provincial Red- and Blue-listed species have no legal status, although a few species are

[1] Numbers updated since paper delivered.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

included in the Identified Wildlife Management Strategy (IWMS), Volume 1.

The British Columbia government has stated emphatically that it will not pass any separate endangered species legislation. The Minister of Environment, Lands and Parks has declared that the current legislation, particularly the Ecological Reserve Act, the Park Act, the Wildlife Act, the Forest Act, the Forest Practices Code, the Environment and Land Use Act, the Land Act, and, more recently, the Fish Protection Act and the Identified Wildlife Management Strategy, will protect species at risk.

Unfortunately, this is not the case. For example, the IWMS has serious shortcomings, clearly identified by the British Columbia Endangered Species Coalition (1998). It lacks real habitat protection, and even first-year biology students know that you can't save species without saving habitat. The British Columbia government has directed that impacts resulting from the application of the IWMS (for both Volume 1 and Volume 2 species) cannot exceed a maximum of 1% impact on the 1995 annual allowable cut. Only 34 CEO-approved wildlife species and 4 plant communities are included in Volume 1. Yet there are more than 1,575 species at risk in British Columbia. At this time only interim measures can be applied for the marbled murrelet, an old-growth specialist requiring specific habitat types within the Coastal Western Hemlock ecosystem. Such specialized landscape units may not be easily saved in future, because in current Landscape Unit Objectives representativeness at a scale finer than the Biogeoclimatic Ecosystem Classification variant (Meidinger and Pojar 1991) will not be adopted, in spite of the recommendations in the Forest Practices Code Biodiversity Guidebook.

Most of the species at risk in the province occur in the Southern Interior and Georgia Depression ecoprovinces, in ecosystems already listed as endangered. In both of these ecoprovinces there has been a dramatic loss of habitat because of urban and agricultural development, and most of the remaining nonforested land is in private ownership. Under these circumstances, the Forest Practices Code, the Identified Wildlife Management Strategy, and the other government acts provide little protection.

British Columbia has signed the National Accord for the Protection of Species at Risk. The auditor general has documented that the federal government has done little concrete in the area of environmental protection and protection of species at risk. Bill C-65 "An Act respecting the protection of wildlife species in Canada from extirpation or extinction," which received first reading on 31 October 1996, had serious deficiencies that were well documented by the Sierra Legal Defense Fund, the Canadian Endangered Species Coalition, and more than 300 scientists who wrote twice to the prime minister and the federal minister of the environment. The listing process was unacceptable, there was no real habitat protection, and there were problems with transboundary species. Hundreds of scientists across the country continue to stress these deficiencies. Fortunately, Bill C-65 died on the order paper in the House of Commons after the 1997 election was called. There is serious concern that the new endangered species act now being drafted by the government of Canada will be even weaker than Bill C-65.

Meanwhile, Bill C-441, with the same title as Bill C-65, has been introduced as a private member bill by Charles Caccia, Liberal MP for Davenport, and chair of the Standing Committee on Environment and Sustainable Development, to fill the void. It received first reading on 8 October 1998, and is a model bill that has received high praise, namely an "A" grade from the major environmental groups. Because habitat loss is the primary threat to over 80% of Canada's species at risk, habitat protection is the key to saving species. Bill C-441 recognizes this and would protect the habitat of all endangered species on all lands. If the new federal government bill being drafted mimics Bill C-441 we could be well on the way to saving our species at risk. This seems highly unlikely, and so harmonization across the country would do little for species at risk.

CONSERVATION STRATEGIES

In British Columbia, only 1 species, namely the Vancouver Island marmot, has land set aside under the provincial Wildlife Act to specifically protect a species at risk. For the most part, the province's commitment to saving species at risk and conserving biodiversity depends on a planning framework that accommodates 2 complementary components: a network of protected areas, and the application of integrated resource management principles outside these protected areas (B.C. Ministry of Environment, Lands and Parks and Ministry of Forests 1992).

The Protected Areas Strategy (PAS) will protect 12% of the province as a whole by the year 2000 (British Columbia 1993*a*). To this end, the government has moved from having 6.3% (5.95 million ha) of the land base dedicated to protected areas in 1990, to approximately 10.6% (10.06 million ha) by 1998 (SOE 1998). There seems little doubt that British Columbia will achieve the 12% goal by the year 2000, but there will be over-representation of higher-elevation ecosystems, and under-representation of lower-elevation and valley-bottom ecosystems, especially in the southern Interior (Pojar 1996).

In British Columbia, protected areas include national parks, ecological reserves, class A and C parks, recreational areas, and other areas that fall under the Environment and Land Use Act (SOE 1998). They do not include wildlife reserves, migratory bird sanctuaries, and regional parks (SOE 1998), although these have some conservation value. In the 1930s and 1940s parks were set aside primarily to encourage

tourism, but by the 1970s and 1980s park creation began to focus on protection of unique natural environments (SOE 1998). In the 1990s, representation of British Columbia's biological and cultural diversity, recreational resources, and habitat protection became the primary objectives (SOE 1998).

The first goal of the PAS is to protect viable representatives of natural diversity (British Columbia 1993*a*). For the purpose of the PAS, the measurement of diversity is limited to an assessment of an area's richness, and this is richness as it is applied to natural, cultural heritage, and recreational values. In other words, diversity in this context is not the same as biological diversity (Scudder 1996*b*).

The second goal of the PAS is to protect, among other things, rare and endangered species and biologically exceptional sites—the latter being defined as to include the sites of high species richness, sites with endemic species, and sites of species at the extremes of their range (British Columbia 1993*a*).

The map of our protected areas in the province shows a scattered pattern of distribution involving both small and some significant large areas. It might appear that these areas are well situated to protect our species at risk. Detailed analysis shows that this is not the case. The recent mapping by the British Columbia Endangered Species Coalition shows that it is not so on a large scale and not even at a fine scale. In this mapping, based upon data from the British Columbia Conservation Data Centre, the wide-ranging large carnivores and ungulates are excluded and multiple hits at a site are obscured.

The fact that our protected areas seem not well placed for protecting our species at risk is not surprising given the differing strategies used over the years to select these parks and other areas now classified protected areas. It is clear that the protected areas are insufficient to save our species at risk.

A major problem exists in our current conservation strategies, in that these have generally lumped together strategies for conserving rarity and strategies for conserving richness within the 12% objective. These efforts are based on inadequate data, often with misplaced emphasis on vertebrates.

The Gap Analysis Workbook (British Columbia 1993*b*) notes that limited knowledge is available about most species, and that there are limited resources available to obtain the information required. However, it is important to acknowledge that over the past few years, through various Forest Renewal British Columbia (FRBC) programs, there have been some admirable new initiatives in this area; some with very significant results. Unfortunately, we must also note that FRBC has announced that it will no longer fund research on sensitive wildlife species (Pynn 1999).

Species richness measures generally focus on better-known taxa, particularly vertebrates and vascular plants (Reid et al. 1993). On the assumption that vertebrates provide an adequate umbrella for invertebrates at most levels (Murphy and Wilcox 1986), it is often postulated that vertebrate species-richness can be used as an indicator of overall natural diversity that integrates a host of community-related parameters (Scott et al. 1987). It is also assumed that areas with high species-richness in vertebrates and vascular plants tend to have high species-richness in other groups (Reid et al. 1993). This is despite knowledge that shows that, whereas many vertebrates are wide-ranging generalists, many invertebrates are characterized by restricted distributions, movements, and associations with unique habitats (Hafernik 1992).

A study of the occurrence of high species-rich areas in Britain shows that the species-rich areas of different biota do not coincide (Prendergast et al. 1993). Using distributional data sets produced from the fine scale and high intensity of recording in Britain, these authors defined species-rich "hot spots" as the top 5% of record-containing 10-km squares (ranked by number of species per 10-km square), and showed that there was little coincidence of different taxa richness "hot spots." They showed that the maximal overlap (34%) was between butterflies and dragonflies, and that there was only 12% overlap between 2 of the "flagship" conservation taxa, namely birds and butterflies. In a similar study in Uganda, Howard et al. (1998) also found little congruence in the species richness of woody plants, large moths, butterflies, birds, and small mammals.

As I noted in *The Wilderness Vision for British Columbia* (Scudder 1996*b*), to date we have not started to address this problem in British Columbia, although we could do this given clear objectives, cooperation between agencies, and sufficient funding. Good distributional data are available for most of the vertebrate groups and probably for many of the plant taxa, and recently we have developed a geo-referenced database for the distribution of our dragonflies, butterflies, and true bugs. Until the richness "hot spots" are scientifically determined for the different biotic taxa, it will not be possible to clearly define the conservation objectives, nor work out the appropriate conservation strategies for species richness.

It is important to determine if the richness "hot spots" in British Columbia coincide with the species at risk "hot spots." Although there appears to be some coincidence, as judged from our recent biodiversity assessment in the Montane Cordillera ecozone (Scudder and Smith 1999), it would be surprising if there is a good overlap across the whole province. There are now a number of studies that show that richness "hot spots" do not usually coincide with rarity "hot spots." Prendergast et al. (1993) have shown that this is the case in Britain, and Curnutt et al. (1994) have also documented this in Australian birds.

Prendergast et al. (1993) defined rare species as those that occurred in less than 16 of the 2,761 10-km squares in their British database, and found that there was a lack of coincidence. About half (43%) of the rare bird species were not found in bird "hot spots." Similarly, Curnutt et al.

(1994) found that rarity and richness "hot spots" did not coincide, but found that rare species contribute more to bird richness "hot spots" in Australia than they do in Britain. They point out that targeting areas of high diversity may be the best way to protect rare species only if very large areas are available for conservation, a view endorsed by Reid (1998). Unfortunately, this strategy is unlikely to help in British Columbia because in the relevant ecosystems in the Southern Interior and Georgia Depression ecoprovinces, there are no large areas left for conservation purposes.

Curnutt et al.'s (1994) study did not consider the problem that exists over the lack of coincidence of rarity "hot spots" in the different taxa. This problem has been well documented by Dobson et al. (1997) in their study of the distribution of endangered species in the United States. They showed the rarity "hot spots" for different groups rarely overlap. This is also likely to be the case in British Columbia, except perhaps in the Queen Charlotte Islands, the South Okanagan, the Georgia Depression, and the Peace River Basin.

So, what do we do? The PAS provides no acceptable conservation strategies for this situation. The conflict over conservation of rarity versus richness "hot spots" must be resolved. In my opinion, our present strategies are largely ad hoc, unintegrated, and ineffective across the province, driven more by politics than science. Instead, we need to adopt efficient, proven methods for choosing priority areas for conservation—methods that can consider both species at risk and biodiversity-rich areas.

Setting priorities for conservation is unavoidable. Competition for land limits the areas available, so conservation goals have to be established and obtainable within these limits (Pressey 1994, Pressey and Tulley 1994).

Our conservation strategies need to be revised and revitalized. We must first establish our conservation objectives, which will obviously include saving our species at risk. However, we also want to save other species and develop a functional conservation network. Further, since some habitats have already been set aside in protected areas, these should be incorporated into the network, where possible, and added to in the most effective way (Nicholls and Margules 1993). Finally, it is essential to ascertain the minimal area needed to achieve these objectives, while at the same time be able to test alternatives and adjust to changing priorities (Pressey et al. 1994). This is a tall order, but all of this could be done if a new approach to selecting conservation areas were adopted using some key principles for systematic conservation area selection.

The 3 principles relevant to this strategy are irreplaceability, complementarity, and flexibility (Pressey et al. 1993). Use of such principles distinguishes between irreplaceable and flexible areas, and results in very different conservation priority areas (Williams et al. 1996). GIS-based heuristic algorithm methodology has the ability to incorporate the views of community groups and government agencies, and to identify negotiable and nonnegotiable options (Pressey et al. 1995). Although it may not guarantee an optimal solution (Pressey et al. 1996), it provides conservation biologists with a definable, scientifically based conservation strategy.

Complementarity solutions have been developed for American endangered species (Dobson et al. 1997), and show that the amount of land needed to cover the endangered and threatened species in the United States may be a relatively small proportion of the land mass. Ongoing research is already showing that a similar situation prevails applying irreplaceability and complementarity solutions for COSEWIC-listed species in Canada (Freemark et al. 2000).

A similar analysis needs to be done for the Red-listed and Blue-listed species in British Columbia as a whole. Warman (2000) is already doing such an irreplaceability/complementarity analysis for the species at risk in the South Okanagan, and Forsyth and Sinclair (2000) are conducting irreplaceability research on the British Columbia lake fish fauna.

The data are available to do such analyses for the terrestrial biota for the whole province. The methodology is here. The expertise is here. Students at the University of British Columbia are now working closely with the Australian software developers, who have demonstrated many varied applications of this program. We have the will, but we don't have the funding in spite of our continued attempts to obtain it.

However, simply determining such a minimal set of priority areas, and seeing what extra lands need to be identified for conservation is not enough. The irreplaceability/complementarity methodology will not fully be able to select the best areas for conservation until the results are integrated with some of the important factors affecting viability, threat, cost, and management options (Williams et al. 1996).

Without an appropriate landscape context to allow metapopulation dynamics, species at risk will not be saved. A province-wide, integrated conservation network with core areas is needed, at least on an ecoregion scale (Noss 1992). Some attempt at this has been made in provincial land use planning, but only on a local scale (e.g., Vancouver Island, Cariboo–Chilcotin). These did not solve major conservation concerns; for example, the Garry oak ecosystem on Vancouver Island was excluded because little is on Crown lands. Currently Land and Resource Management Plan (LRMP) decisions are not well integrated and certainly do not have conservation strategies as the main objective.

The proposed state-wide network for Florida (Noss 1985, 1987, 1992) indicates the scale and scope of such planning needs. It also demonstrates what likely needs to be done to save large carnivores at risk, a fact reiterated in carnivore conservation studies in the Great Lakes region (Mladenoff et al. 1997) and the Rocky Mountains (Noss et al. 1996). Large carnivores can serve as useful umbrella species, because their conservation requirements are likely to encompass

those of many other components of communities owing to their large space requirements (Schonewald-Cox and Buechner 1991), but they cannot be the sole focus (Noss 1992). For example, Flather et al. (1998) point out that grizzly bear range fails to overlap with the major species endangerment regions in the United States. From a simple inspection of the current distribution of the grizzly bear in British Columbia and examination of the British Columbia Grizzly Bear Conservation Strategy (B.C. MELP 1995) it is obvious that this species cannot serve as an umbrella species for the South Okanagan and Lower Similkameen. For example, the Mormon metalmark butterfly is confined to a few locations in the bottom of the Similkameen Valley near Keremeos (Scudder 1994, 1996*a*, Guppy et al. 1995), and is dependent on buckwheat for larvae feeding, in association with flowering rabbit-brush for adult nectar feeding—a life cycle very different from that of the grizzly bear.

For every species at risk, at least a minimal viable population and the metapopulation must be maintained, although these are not always easy to calculate and determine (Thomas 1990). A population vulnerability analysis (PVA) must be undertaken involving assessments of the population biology, namely the population phenotype, environment, and population structure and fitness (Gilpin and Soulé 1986). Although these analyses need to be done with caution (Beissinger and Westphal 1998), we should have a PVA for all of our species at risk.

If these assessments are not done correctly and do not incorporate what Hendrick et al. (1996) call an inclusive population viability analysis, basic conservation biology principles dictate that species at risk are liable to be caught in what Gilpin and Soulé (1986) describe as the demographic, the fragmentation, the inbreeding, or the adaptation vortices. Any 1 of these vortices can lead to species extinction or extirpation.

Species differ in their vulnerability to the 4 vortices (Gilpin and Soulé 1986), although some predictions can be made. For example, *r*-selected species, notably insects, fish, and rodents (especially those with good long-range dispersal), are unlikely to suffer from the inbreeding or adaptation vortices. Usually their total population sizes and rates of gene flow are enough to avoid the loss of genetic variation. However, even butterflies can be victims of the inbreeding vortex (Saccheri et al. 1998), and local populations of insects can get caught in the demographic vortex as shown by Ehrlich (1983) in his research on Edith's checkerspot butterflies. Usually in wild populations, demographic factors are of more importance than genetic factors in species at risk (Lande 1988).

Large vertebrates can persist at low population levels for some time because of longevity and because they are buffered from short-term environmental changes, but have a high probability of entering the inbreeding vortex (Gilpin and Soulé 1986). Territorial species are a special concern (Lande 1987). No single conservation strategy fits all.

Our overall conservation strategy should be to establish areas large enough to contain all of the species at risk, then maintain and manage them to assure their persistence (Harris 1993). However, small reserves continue to fill a worthy niche in conservation strategies (Shafer 1995).

ECOSYSTEM INTEGRITY

Just setting aside protected areas is not enough even if they are large. Newmark (1995) has analyzed large mammal extinctions in western North American parks and shows that extinctions in protected areas are a fact of life, and extinction rates are higher in smaller reserves. Many of British Columbia's protected areas do not have buffer zones, and few are large enough to accommodate the species at risk they profess to conserve. Even surrounding protected areas by buffer zones and special management zones may not suffice, especially if the ecosystem containing them lacks integrity.

Habitats, protected areas, buffer zones, and special management zones are contained within ecosystems, and to save them in the long term we must save these ecosystems. This is now the rationale behind the ecosystem RENEW strategy in the South Okanagan, the continuation of the earlier biodiversity inventory (Harper et al. 1993) and South Okanagan Conservation Strategy (Illady 1993).

Saving ecosystems means that ecosystems must retain their integrity (Angermeier and Karr 1994), that is, their composition and function. Ecosystems are complex and their precise spatial and temporal boundaries are often difficult to define. As complex systems, ecosystems have a number of important properties (Kay 1991, 1993). These properties include:

1. non-linear: the system behaves as a whole; it cannot be decomposed into pieces and then reconstructed by adding the pieces together;
2. multi-scaled: cannot be understood by focusing on 1 scale alone;
3. window of vitality: must have enough complexity, but not too much; complex systems strive to stay within a "window" of vitality;
4. multiple steady states: they have no preferred unique steady state;
5. dynamically stable: equilibrium points for the ecosystem may not exist;
6. catastrophic behaviour: can have sudden, dramatic changes; flip-flops;
7. informational libraries: contain genes.

Ecosystems are thermodynamically open systems, which are never in complete equilibrium. They are open because they take in energy from the sun.

Ecosystems are also self-organizing and, as such,

characteristically may undergo abrupt changes when a new set of interactions and activities emerge among components and in the whole system. The form that this self-organization takes is not predictable in advance. This is because the very process of self-organization is sudden and one can get "flips" into a new regime. One of the characteristics of sudden change is that ecosystems may have several possible behavioural pathways available to each "flip." Thus, in a specific geographical location there is the potential for a number of different ecosystems, communities, and species to exist in addition to the present one. For example, much of our forested land here in British Columbia (given sufficiently heavy-handed use) could be ericaceous heath (Kimmins 1996*a*). Which pathway is followed is often, but not always, an accident of circumstances.

Once formed, an ecosystem has a certain amount of integrity. That is, it is able to successfully survive in spite of various perturbations. In other words, it has a certain self-organizing or self-stabilizing ability. However, there are limits to this ability, and at any time it could move beyond its normal stable state (Graham and Grimm 1990, Kimmins 1996*b*).

Ecosystems are not in a permanent stable state. They are continually undergoing a birth, growth, maturity, and death process. Holling (1986) refers to this as the exploitation, conservation, release, and reorganization cycle, depicted as a figure-8 (Fig. 1). Essential components of this cycle are natural disturbance events like individual tree-fall in tropical rain forests (Hubbell et al. 1999), windthrow in humid coastal forests in British Columbia, and fire and insect pest outbreaks in the interior and boreal forest (Kimmins 1996*a*).

Disturbance is a vital process in ecosystems, affecting biological diversity and ecological function (Loucks 1970, Pickett et al. 1989, Attiwill 1994*a*,*b*). As humans alter natural disturbance regimes, disturbance-driven ecosystems and disturbance-dependent ecosystem conditions will change (Kimmins 1996*a*).

On the local scale, movement through the "figure-8" ecosystem cyclic process is not continuous (Holling 1986). There can be temporary stationary phases. Each of these temporary stationary phases is very sensitive to certain sets of conditions. Unfortunately, it is not possible to predict what these conditions might be.

We know that in the conservation or mature state (what used to be called the climax state), most of the nutrients and energy are locked up in the biomass, and the system gradually becomes "brittle." Key structural parts become risk-prone, waiting for an "accident" to happen (Holling 1986).

All of these "accidents" can lead to the relatively fast, downward process of release; a process that usually occurs in patches, providing the ecosystem or community is large enough to accommodate patchiness. Ecological integrity requires patchiness. For functional patchiness to occur, large areas are often needed. There are very few calculations on how large an area has to be in order to maintain ecological function and integrity over the long term. Shugart and West (1981), working on long-term dynamics of forested ecosystems, estimated that the landscape must be some 50–100 times larger than the average natural disturbance patches in order to maintain a relatively steady-state ecosystem. Recently, He and Mladenoff (1999) have carried out a spatially explicit and stochastic simulation of forest landscape fire disturbance and succession in a heterogeneous landscape in northern Wisconsin, and have demonstrated the complex interactions that can occur over a range of temporal and spatial scales.

There are no estimates of how much of an ecosystem is needed to retain its integrity and sustainability. It seems likely that this cannot be done with less than 50% of an ecosystem, especially when natural disturbance regimes no longer prevail. Human-dominated ecosystems are the norm in the world today (Noble and Dirzo 1997, Vitousek et al. 1997), with fire suppression, livestock grazing, unnatural fragmentation, alien species introduction, and other unnatural disturbances a consequence. It is recommended that to maintain biodiversity, forest practices should mimic natural disturbance patterns (Bunnell 1995). These differ across forest types, and although the British Columbia Forest Practices Code specifications now try to mimic natural disturbance regimes, they do so without them being fully known. Andison

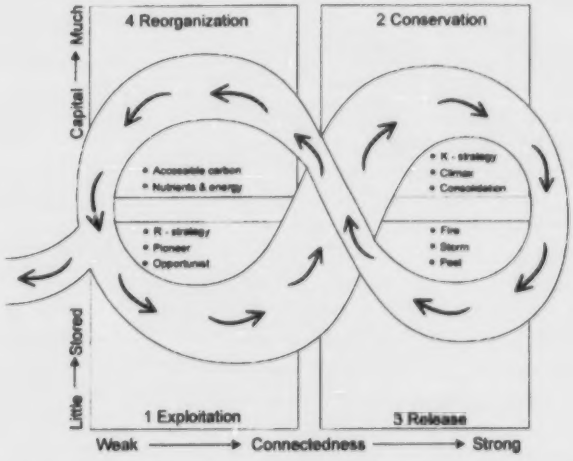

Figure 1. The "figure-8" model of an ecosystem (redrawn after Holling 1986). The arrows show the speed of the flow of events in the ecosystem cycle, where arrows close to each other indicate fast change and arrows far from each other indicate slow change. The exit for the cycle at the left of the figure indicates the stage where a flip into a greatly modified ecosystem is most likely.

(1996) has documented the consequences of applying the new code in the Prince George area; it will not result in a natural disturbance landscape.

If a mistake is made, an ecosystem could well flip into another stable state, a state very different to that now needed to maintain our species at risk. Mankind has caused such flips elsewhere in the world in the past, so this isn't science fiction. Flips can be seen to have taken place in the Mediterranean (Thirgood 1981), the New Guinea highlands and the Scottish moors, and are likely in the Amazon (Shukla et al. 1990), with loss of biodiversity as a consequence. A recent food-chain flip encompassing species that are listed as at risk, involves killer whale predation on sea otters (Estes et al. 1998), although the causes may be complicated (Kaiser 1998), and not just the anthrogenic changes in the offshore oceanic ecosystem as suggested by Estes et al. (1998).

To save our species at risk, ecosystems must be saved and allowed to function as a whole in a natural manner, that is, retain integrity. At present our planning is not on this scale. These ecosystems are already being impacted by new stressors that cannot be controlled. These include rising greenhouse gas concentrations and disruption of the natural ozone regime (Fergusson and Wardle 1998), with every indication that these perturbations will intensify over the next few decades (Schindell et al. 1998).

Since there is a dramatic CO_2 increase in the atmosphere (Schindell et al. 1998), and the Vostok ice cores from the Antarctic show a close correlation between CO_2 and temperature (Raynaud et al. 1993), most atmospheric scientists now accept that increased greenhouse gases will lead to global warming (Harrington 1987, Houghton and Woodwell 1989, Schneider 1989, McBean and McEwan 1990). Although there are some differences between the various General Circulation Models, they all reveal that there will be an increase in global temperature, due to CO_2 doubling, of 3.5–5.2°C, and that the increase in the earth's surface temperature will be greatest in the higher latitudes (Wetherald 1991). Accumulated temperature changes in Canada from 1993 to 1997, relative to 1951–1980 averages, show a definite increase in the western Arctic (Fergusson and Wardle 1998).

British Columbia can expect the average temperature to be 2–4°C higher in June to August and 4–6°C higher in December to February (Hengeveld 1989, Canadian Climate Program Board 1991). A 3°C-warming would present our ecosystems with a warmer world than has been experienced in the past 100,000 years (Schneider and Louder 1984), while a 4°C-rise would make the earth the warmest since the Eocene, some 40 million years ago (Barron 1985, Webb 1992).

If the present rate of CO_2 accumulation continues, and this seems likely according to the Goddard Institute (GISS) model (Schindell et al. 1998), doubling of CO_2 appears possible as early as 2030 AD, and highly probable by 2050 AD (Bolin et al. 1986, Schindell et al. 1998). That means it may be only 31 years from now!

Such a warming trend would not only be large compared to natural fluctuations in the recent past, it could also be 10–100 times faster than long-term natural average rates of change (Schneider et al. 1992). Such a rate of change may exceed the ability of most species to adapt (Peters 1991), and will have repercussions throughout all of our ecosystems (Melillo 1999). There will be changes in the grasslands and the forests (Pitelka et al. 1997, Alward et al. 1999).

This climate warming will have a very large impact on our biodiversity and species at risk, because it is sure to change both ecosystem structure and function (Peters 1991). There will be species movement and ecosystem community disruption at least comparable to what has happened in the past (Webb 1981, Wolfe 1985, Delcourt and Delcourt 1987, FAUNMAP Working Group 1996, Bennett 1998, Cain et al. 1998): ecosystems as a whole do not move. Species will respond to climate change individually, and with a 3°C-increase we can expect a 250-km latitude change and/or a 500-m altitude change in their range as a minimum (McArthur 1972, Dorf 1976, Furley et al. 1983). Such range movements are already suspected in insects (Parmesan 1996).

This has serious implications for our conservation planning because our species at risk may no longer remain in reserves and other areas set aside for them (Peters and Darling 1985). To save our species at risk in this context, there must be appropriate landscape continuity and corridors, or we may have to move our protected areas. All species at risk, with their very different means and modes of dispersal (Hansson et al. 1992) must be accommodated.

This is conservation planning on a scale not contemplated now, a scale that crosses administrative boundaries, as well as political and international borders. It is a scale already adopted in attempts to save the grizzly bear in the Yellowstone to Yukon conservation initiative (Locke 1998), and calls for international cooperation and legal agreements (Keiter and Locke 1996).

The present LRMP process, while locally relevant for the short term, is on a wrong scale for these long-term considerations. If today's problems in obtaining agreement at the various LRMP tables are a guide, imagine the task now at hand. Politicians will hate it, local communities will not understand it, and big business will be sure to oppose it. Yet, it is essential if species at risk are to be saved within the usually accepted 100- to 500-year conservation horizon, a horizon that deals with our grandchildren and our grandchildren's grandchildren.

Politicians must scrap the self-imposed 12% protected areas maximum, which has no scientific validity. Industry's short-term demands should no longer be acceptable. Large-scale landscape conservation planning in an ecosystem context must be adopted, wherein at least 50% of the area of each of

Table 1. Ecosystem Services[a] (after Costanza et al. 1997).

Gas regulation	Pollination
Climate regulation	Biological control
Disturbance regulation	Refugia
Water regulation	Food production
Water supply	Raw materials
Erosion control and sediment retention	Genetic resources
Soil formation	Recreation
Nutrient cycling	Cultural
Waste treatment	

[a] Includes ecosystem "goods" along with ecosystem services.

our land-based ecosystems is managed for maintenance of their ecosystem integrity. Yet economic viability must be sustained and the ever-increasing population accommodated.

Can it be done? I don't know, but conservation biologists must try. The fact that we are here, talking about species at risk, means that we have not made much progress. At this time we can't even get endangered species legislation established, and that is the easy part.

I don't think these conservation objectives can be achieved by relying on appeals to save species at risk or biodiversity in general. These issues do not sell to anyone except the converted.

I have stressed the ecosystem integrity context and believe this is a much more saleable point if we stress ecosystem services (Table 1). These ecosystem services are invaluable (Daily 1997) and readily appreciated. Mention clean air and clean water, and everyone is interested, since their survival depends on them.

We will not achieve the conservation objectives by giving scientific lectures to the converted. It is the general public that needs to understand the conservation concerns and objectives. Biologists must go to all sectors of society (Powledge 1998), explain the options, and then work to facilitate locally acceptable solutions within well-defined conservation objectives. Objectives that are scientifically sound, long term, ecosystem based, and well integrated. We must work with the aid of GIS-based land use planning tools that have the capacity to provide both flexibility and continuous reassessment. The public must be drawn into our planning process.

The South Okanagan, with so many species at risk, is an ideal place to try these new conservation strategies. A group put together by K. Freemark from the Canadian Wildlife Service in Ottawa along with D. Olson of Olson + Olson Planning and Design Consultants, a Calgary environmental planning and landscape architecture firm, has now embarked on an "Integrated Landscape Planning and Assessment" study in this highly impacted area.

Relatively speaking, the scientific part of this program is easy or, at least, the area with which most of us are already involved and comfortable. Government and industry buy-in is a must. Public involvement is essential. Education is imperative. Such a multifaceted effort in the South Okanagan is underway. It is a beginning.

In order to succeed, we must get all members of society to respect nature—respect and value the benefits that the natural world provides us. Only then will we save species at risk.

ACKNOWLEDGEMENTS

Research for this paper was supported by grants from the Natural Sciences and Engineering Research Council of Canada. W. Klenner and J. Pojar provided useful comments on the paper, but are in no way responsible for any of the statements contained therein. I thank my daughter, N. Scudder, for typing the early draft of the manuscript and for improving the text. L. Lucas kindly did the final processing and preparation of Table 1 and Figure 1.

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The Conservation Data Centre: For the Greatest About the Least

Andrew P. Harcombe
Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
Victoria, BC, V8W 9M1, Canada
Andrew.Harcombe@gems2.gov.bc.ca

ABSTRACT

Key concepts and terms such as elements at risk, Red and Blue Lists, and ranking are defined to facilitate effective communication in other presentations. How the lists are created and by whom is also explained. The British Columbia Conservation Data Centre (CDC) is a centralized, computer-assisted inventory and information system about rare and endangered species and natural communities (plant associations). The CDC has been in operation for 9 years, is part of a large network of centres (6 others of which are in Canada), and uses a common methodology to ensure data sharing and potential information integration. Many government and nongovernment initiatives that have been instigated since the CDC began have required organized information on the province's rare elements of biodiversity, and we have been there to help. The breadth of our services has increased since those early days; this paper is intended to update you on these services and our plans for the coming years. Included will be a brief discussion on related extra-provincial activities.

Key words: Blue List, Conservation Data Centre, Red List.

DEFINITIONS AND STANDARDS

One of the keys to successful communication is a common understanding of terminology for the subject area. Over the years, various organizations have used a wealth of terms to describe rarity of organisms and ecosystems. These terms have ranged from very subjective descriptions to precise indices derived from complicated algorithms. Because of this overlap between informal and formal use of terms such as "endangered," I propose some solutions. To bring some standardization to the other papers, a glossary of terms is given in Table 1. The greatest problem probably arises when directly comparing different ranking systems (Pearl 1997). Not only do the same terms differ in definition, but the rarity classes do not directly correspond to each other. For example, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) boundary for degree of "at risk" between Threatened and Vulnerable does not match the International Union for Conservation of Nature and Natural Resources (IUCN) boundary between Vulnerable and Lower Risk.

THE CONSERVATION DATA CENTRE

The last major conference in British Columbia about provincial biodiversity was *The Living Legacy* in 1991. At that time, the Conservation Data Centre (CDC) was only weeks old and starting out on an ambitious plan to compile and analyze information on the rare vascular plants, vertebrates, and forested ecosystems of the province. I reported what the CDC was going to do (Harcombe 1991). Now, 9 years later, I can report on what has happened, as well as the plans for the coming years. The CDC has grown in staff, scope, data, expertise, and demand. Thirteen specialists now work in botany, zoology, ecology, and data management. The CDC responds to >1,000 data requests per year, and through its Web page, makes information available to many more. We have tracking lists (lists of elements for which we actively compile element occurrence data) for vertebrates, forests, vascular plants, nonforested ecosystems (currently under expansion), bryophytes, lichens, marine seaweeds, big trees, and selected invertebrates (freshwater mollusks, dragonflies and damselflies, butterflies, robber flies, and marine benthic invertebrates). Our specialists regularly provide expert advice, and are especially valuable in those areas of wildlife and ecology where traditional knowledge is not as widespread. Besides our own professional specialists, we have incorporated the expertise of others, such as Trevor Goward, Hans Roemer, Frank Loomer, Bill Austin, the late Gerald Straley, Del Meidinger, Jim Pojar, Will McKenzie, Jon Shepard, Cris Guppy, and Michael Hawkes.

Over the past year, the CDC has produced 4 major publications on rare elements: native vascular plants (Douglas

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1. Definitions for rarity and related terminology.

at risk	elements that can be either at risk of extinction, defined in terms of the probability of extinction within a specified time period; or, elements at risk because population viability is not considered to be secure in British Columbia at this time
Blue List	vulnerable taxa that could become candidates for the Red List in the foreseeable future; taxa generally suspected to be vulnerable because information is too limited to allow designation in another category
COSEWIC	Committee on the Status of Endangered Wildlife in Canada; Canada's official body for determining national status of indigenous taxa and distinct populations
designation	official or formal assignment of an "at risk" class to an element; usually only occurs after a detailed status report is prepared (usually includes recent field data gathering) and peer-reviewed; usually assigned by legislation
ecosystems	a unit of land or water comprising populations of organisms considered together with their physical environment and interacting processes between them
element	for the CDC, a definable component of biological diversity, including species, subspecies, variety, genetically distinct population, or ecosystem
occurrence	a geographical location used predictably by an element for a specified purpose, such as breeding, feeding, staging, or resting. For plants, all populations of a species are occurrences; for animals, many observations of an element do not constitute a predictable use of a habitat or habitat element and may not be occurrences.
endangered	any indigenous species that, based on the best available scientific evidence, is indicated to be imperilled with imminent extirpation or extinction throughout all or a significant portion of its British Columbia range. The term is also used as an official legal designation of Endangered.
extant	now in existence
extinct	species that no longer exists anywhere
extirpated	species that were once part of the natural flora and fauna of British Columbia and no longer occur here but occur in captivity or in the wild elsewhere
historic	usually not verified for some time (25–40+ years); not enough up-to-date inventory is available that would remove the expectation that the taxon someday may be rediscovered. Once it is considered gone, it would be termed extirpated.
IUCN	International Union for Conservation of Nature and Natural Resources; their Species Survival Commission assesses the status of species, subspecies, populations, and stocks on a global scale to determine species at risk of extinction and promote their conservation
ranking	assessment of rarity initially based on professional evaluation that may lack the detail that would allow more formal evaluation without further field inventory
Red List	taxa have either very small local populations or biological characteristics that imperil them. This includes species or subspecies designated as Threatened or Endangered under the British Columbia Wildlife Act; species or subspecies that are candidates for legal designation as Endangered or Threatened; or species or subspecies that have been extirpated but were once part of the natural flora and fauna of British Columbia.
status	may be a formal or informal evaluation of the state of health of a species or ecosystem; in Canada, this is both done provincially by various groups and nationally by the Committee on Status of Endangered Wildlife in Canada (COSEWIC)
taxon (plural: taxa)	formally named, related group of organisms at any level in a classification (e.g., family, species, and subspecies)
threatened	any indigenous species that are likely to become endangered if the factors affecting population viability are not changed. This species, based on the best available scientific evidence, is indicated to be experiencing a definite noncyclical decline throughout all or a major portion of its British Columbia range, or any species with an extremely restricted distribution in a habitat with a high probability of environmental degradation. The term is also used as an official legal designation of Threatened. The IUCN uses the term to refer to its 3 classes of rarity: Critically Endangered, Endangered, and Vulnerable.
vulnerable/ sensitive	any indigenous species that is particularly at risk in British Columbia because of low or possibly declining populations; likely to move into the threatened category in the near future if causal factors continue operating. The IUCN uses the term as an official designation of Vulnerable.
Yellow List	taxa that are considered to be secure in British Columbia now; they are managed at the habitat level by managing for a diversity of habitats in the province
wildlife	flora and fauna of a region

Table 2. Conservation Data Centre ranking system.

1	Critically imperilled because of extreme rarity (5 extant occurrences or very few remaining individuals) or because of some factor(s) making it especially vulnerable to extirpation or extinction.
2	Imperilled because of rarity (typically 6–20 extant occurrences or few remaining individuals) or because of some factor(s) making it vulnerable to extirpation or extinction.
3	Rare or uncommon (typically 21–100 occurrences); may be susceptible to large-scale disturbances (e.g., may have lost extensive peripheral populations).
4	Frequent to common (>100 occurrences); apparently secure but may have a restricted distribution; or there may be perceived future threats.
5	Common to very common; demonstrably secure and essentially ineradicable under present conditions.
	Qualifiers
A	An element (usually an animal) that is considered accidental or casual in province; a species that does not appear annually
B	Breeding; the associated rank refers to breeding occurrences of mobile animals
E	An exotic or introduced species to the province
H	Historical occurrence; usually not verified in the last 40 years, but with the expectation that it someday may be rediscovered, H-ranked elements would typically receive a 1 rank.
N	Nonbreeding; the associated rank refers to nonbreeding occurrences of mobile animals
Q	Taxonomic validity of the element is unclear or in question
R	Reported from the province, but without persuasive documentation for either accepting or rejecting the report (e.g., misidentified specimen).
RF	Reported in error, but this error has persisted in the literature
?	Limited information is available or the number of extant occurrences is estimated
T	Designates a rank associated with a subspecies
U	Status uncertain, often because of low search effort or cryptic nature of the element; uncertainty spans a range of 4 or 5 ranks
X	Apparently extinct or extirpated, without the expectation that it will be rediscovered
Z	Occurs in the province but as a diffuse, usually moving population; difficult or impossible to map static occurrences

et al. 1998*a*), freshwater fish (Cannings and Ptolemy 1998), mammals, reptiles and amphibians (Cannings et al. 1999), and birds (Fraser et al. 1999). Although variable in content, these volumes attempt to synthesize the knowledge of these rare elements in British Columbia. George Douglas has taken the lead role in producing the first 3 volumes of a 7-volume *Illustrated Flora of British Columbia* (1998–1999) (Douglas et al. 1998*b*). We have coordinated writing species accounts for rare ecosystems and vascular plants for potential use in the Identified Wildlife program, and have collaborated on several species/ecosystems at risk brochures (e.g., Flynn 1999; Fontaine and Douglas 1999), one of the most successful wildlife publication series in recent years.

In the coming months, much of our data will become available in the British Columbia (B.C.) Ministry of Environment, Lands and Parks data warehouse, for electronic access both regionally to the Ministry and by Internet for others. Our present data request options will remain in place, either for those without electronic access or for more complex reports and maps. We are in the final stages of implementing geographic information technology, for help with both data capture and user products. This will lead to the availability of some standard map products that will complement our database reports. Sensitive data (as defined by draft Ministry policy) will continue to have location information protected from most viewers; this privilege can be extended to requesters through the regional rare and endangered species specialists (or designates) on a need-to-know basis. The CDC has also become active in conservation evaluation of sites, assessing ecological value for both acquisition and stewardship. This service is of special use to local land trusts and other private conservation organizations. Our specialists' high ranking has encouraged such ongoing property acquisitions such as South Winchelsea Island (The Land Conservancy) and Elkington Garry oaks (Nature Conservancy of Canada).

RANKING ELEMENTS

Each species on the CDC's list is ranked using the system developed over the past 25 years by The Nature Conservancy (U.S.). This system is now used in all Canadian provinces

Table 3. Conservation Data Centre ranking factors.

Known occurrences	A = 0–5 B = 6–20 C = 21–100 D = 101+
Abundance	A = <1,000 individuals B = 1,000–3,000 individuals C = 3,000–10,000 individuals D = >10,000 individuals
Range	A = global: narrow endemic (usually <100 square miles) provincial: very small range, less than 3% of territory B = global: regional endemic (100–10,000 square miles) provincial: narrow range, <10% of territory C = global: moderately widespread, or widespread with spotty distribution provincial: less than half of territory D = global: widespread (>1,000,000 square miles) provincial: more than half of territory
Trend	A = declining rapidly B = declining C = stable D = increasing
Protected occurrence	A = believed to be none protected B = 1 protected C = several protected D = many protected U = unknown
Threats	A = very threatened; species directly exploited or threatened B = moderately threatened; habitat lends itself to alternate use C = not very threatened; habitat is unsuitable for other uses D = unthreatened

and U.S. states, and some Latin American and Caribbean countries. Most government agencies within these jurisdictions have also adopted this ranking system. The CDC uses these ranks to set priorities in tracking the rare and threatened flora and fauna of British Columbia.

Regularly, and I hope at least annually, CDC specialists get together with other species specialists to assess the current status of all elements on the Red and Blue Lists. Any new information on threats, trends, and other criteria is evaluated. Table 2 describes the different rank classes used with a number of modifiers. Table 3 lists the ranking factors considered, with the specific choices for each criteria. Ranking is done at both the global level (G ranks, created by combining information across the network of data centres) and at the provincial level (S ranks).

GLOBAL RANKING

The global (G) rank is based on the status of the species throughout its entire range. The global rank is established by a biologist assigned to that species by The Nature Conservancy (U.S.). The status of a species is ranked on a scale of 1–5 (Table 2). The rank is based primarily on the number of extant occurrences of the species, but other factors such as abundance, trends, range, protection, and threats are also considered if the information is available. To facilitate species rankings, each of the preceding factors is given a letter rank from "A" to "D" (or "U" unknown) for each species, where "A" indicates the greatest rarity or peril, and "D" indicates the least (Table 3). The ranks assigned to particular species are dynamic; as more information is received, ranks may change.

PROVINCIAL RANKING

The provincial or subnational (S) rank is assigned provincially in essentially the same way as the G rank is assigned globally (Tables 2 and 3); the provincial rank cannot exceed the global rank. Generally, the CDC will track only those taxa with provincial ranks of 1–3. Taxa with the ranks of introduced (E) and accidental (A) are not considered eligible for Red and Blue Lists in British Columbia.

The ranking system is related to the Red and Blue Lists as follows:

- Red List: taxa with ranks of 1, 2, 1–2, 1–3, H, or X;
- Blue List: taxa with ranks of 2–3, 3, 3–4; and
- Yellow List: taxa with ranks of 4, 5, or 4–5.

THE FUTURE

For categorizing species at risk in Canada, COSEWIC has adopted slightly modified IUCN criteria; this will be important if federal endangered species legislation is introduced. Because CDCs are important under the National Accord on Endangered Wildlife in providing information regarding species status, we may need to make adjustments to accommodate new information needs. The Nature Conservancy (U.S.) and IUCN are discussing how to better harmonize the systems, which are already very similar. The first *National Status Report on Monitoring the General Status of Wild Species in Canada* is scheduled for preparation by the end of the year 2000 (Mike Oldham, Ont. Natural Heritage Information Centre, Peterborough, Ont., 1999, pers. comm.). A high dependence on the CDC will be required to meet this deadline.

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Going, Gone, and Missing in Action: The Extinct, Extirpated, and Historic Wildlife of British Columbia

David F. Fraser
Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
PO Box 9374, STN PROV GOVT, Victoria, BC, V8W 9M4, Canada
Dave.Fraser@gems8.gov.bc.ca

ABSTRACT

While relatively few species in British Columbia are formally regarded as extinct, there are many species and subspecies that are classified as extirpated and even more listed as "historic", with no verified record in the last 40 years. Fish, birds, mammals, reptiles, vascular and non-vascular plants and invertebrates are listed here: no amphibians in British Columbia are currently on this list.

Key words: extinct, extirpated.

In general, British Columbia has a relatively intact flora and fauna, which is more diverse than any other part of Canada (Harding and McCullum 1994). However, there are a number of species that have been lost from British Columbia and are now extinct or extirpated. An even larger number are "missing"—that is, these species or subspecies have not been recorded in recent times.

The species listed in this paper are extinct, extirpated, or considered "historic." Many of the species in the historic category will likely be relocated once sufficient effort has been spent searching for them. Others, however, require habitats that have largely disappeared, especially those species that were restricted to specific habitats that have been largely lost to urbanization.

For each of these listings, The Nature Conservancy's global ranks (G) and the British Columbia Conservation Data Centre's (CDC) provincial rank (S) are given. In general the ranks run from 1 (critically imperilled) to 5 (secure). The G and S ranks are modified with letters: H = historic, not verified for some time, but may be rediscovered; X = extinct or extirpated, without the expectation that it will be rediscovered; ? = limited information; T = a rank associated with a subspecies or variety. For more detailed information on this ranking scheme in British Columbia see Harcombe (2000), Douglas et al. (1998), Cannings and Ptolemy (1998), Cannings et al. (1999), and Fraser et al. (1999).

These lists frequently change, historic species are found, and new species are added to these lists as records become old enough to meet the criteria for "historic". For the newest listings contact the CDC website at http://www.elp.gov.bc.ca/rib/wis/cdc/tracking.htm.

EXTINCT

The following 5 species and subspecies are considered to be extinct.

Extinct Fish

Dragon Lake Whitefish (Coregonus sp.) GX HX.— This undescribed species differed from *C. clupeaformis* in the number of gill raker counts (Lindsey et al. 1970). The species was eliminated when the lake was "rehabilitated" by poisoning in 1956 (Lindsey et al. 1970, Cannings and Ptolemy 1988).

Hadley Lake Limnetic Stickleback and Hadley Lake Benthic Stickleback (Gasterosteus sp.) GX HX.— Formerly restricted to Hadley Lake on Lasqueti Island. Endemic to British Columbia (McPhail 1988). By 1994 these species were eliminated by the introduction of a nonnative catfish (*Ameirus nebulosus*; Cannings and Ptolemy 1998). The introduced catfish preyed on the eggs of the sticklebacks and probably caused the extinction in a matter of a few years (Hatfield 1999). There are a few other lakes in British Columbia with similar species pairs, and the possibility of the introduction of nonnative fish puts these species at very high risk.

Extinct Birds

Passenger Pigeon (Ectopistes migratorius) GX SX.— Known from a specimen taken at Chiloweyuck (Chilliwack) Depot in June of 1859 and from bones in a midden at Fort D'Epinette in northeastern British Columbia (Campbell et al. 1990). Possibly more numerous in the province than previously believed (Campbell et al. 1990). The last passenger pigeon died in captivity in 1914 in Cincinnati, Ohio.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Extinct Mammals

Dawson's Caribou (Rangifer tarandus dawsonii) GX SX.— Poorly known subspecies restricted to the Queen Charlotte Islands. Last record was of 4 animals seen on Graham Island in 1908, 3 of which were shot (Kelsall 1984, Edwards 1996). Parts of these animals are in the Royal British Columbia Provincial Museum.

EXTIRPATED SPECIES

Extirpated species are those that formerly occurred in the province, and still exist elsewhere.

Extirpated Nonvascular Plants

There are 4 species of nonvascular plants, all mosses, that are regarded as extirpated in British Columbia. Ryan (1996) provides more information on extirpated bryophytes in British Columbia.

Discelium nudum G3G4 SX.— Formerly known from 3 collections on the lower mainland and 1 in the Queen Charlotte Islands. Restricted to earthen banks that are subject to slumping and invasion by vascular plants (Schofield 1990).

Micromitrium tenerum G4 SX.— A moss known in North America only from the Point Grey area of Vancouver (Ryan 1996). Schofield (1990) indicates that these sites have been destroyed by human activity.

Physcomitrium immersum G2G4 SX.— A moss known from single sites in Ladner and Langely at the margins of fields (Ryan 1996).

Pseudephemerum nitidum G? SX.— A moss known in North America from a single site on silt hummocks in a grassy area near the riverbank, MacDonald Beach, Sea Island (Ryan 1996). The habitat has been eliminated as a result of human activity (Schofield 1968).

Extirpated Vascular Plants

There are 4 species of vascular plants that are regarded as extirpated in British Columbia. For more information on extirpated vascular plants see Douglas et al. (1998). All extirpated plant species are on the province's Red List.

Pink Sand Verbena (Abronia umbellata ssp. acutalata) G4TXQ SX.— Formerly found on beaches and sand dunes on the west coast of Vancouver Island (not the east coast as stated in Douglas et al. 1998), south to Oregon. Collections from Pachena Bay, mouth of "Bamfield Creek," and Ahouset (then located in Ahouse Bay, Vargas Island; Ceska 1986). Also extirpated in Washington State (Douglas et al. 1998). The subspecies *acutalata* is the only subspecies found in British Columbia.

Common Downingia (Downingia elegans) G5 SX.— The only known Canadian population was from Leach Lake, near Creston. The muddy shoreline habitat for this species was eliminated during a waterfowl habitat enhancement project (Douglas et al. 1998).

Rabbitbrush Goldenweed (Ericameria bloomeri [Haplopappus bloomeri]) G4 SX.— Formerly known from Westbridge and Spence's Bridge in south-central British Columbia (Douglas et al. 1998).

Lobb's Water-Buttercup (Ranunculus lobbii) G4 SX.— Formerly known from vernal pools and wet sites on southeastern Vancouver Island. Last collected in 1948 in Victoria (Douglas et al. 1998). Older collection sites included Florence Lake, Langford, Glen Lake, Oak Bay, and Blenkinsop Lake (Ceska 1986).

Extirpated Invertebrates

Our knowledge of the rare invertebrates in British Columbia is very poor and this list is likely incomplete. The best known groups, such as butterflies and dragonflies, are being tracked by the CDC. Winchester and co-workers (Winchester and Ring 1996; Winchester 1997*a*,*b*, 1998; Winchester et al. 1999) have argued that the extirpated and extinct arthropod fauna in British Columbia is likely much larger. Scudder (1996) lists a number of species not seen in British Columbia in more than 50 years; however, many of these species would require a specialist's search to relocate them.

Island Marble (Butterfly) (Euchloe ausonides ssp. unnamed) G5TX (G5T1 pending verification) SX.— An undescribed subspecies known only from southeastern Vancouver Island and some of the Gulf Islands. This subspecies is large compared to the other Canadian subspecies, with greener underwings and duskier females (Layberry et al. 1998). Records range from Nanaimo and Gabriola Island in the north, southward to Beacon Hill Park in Victoria (Shepard 1999). In 1998, a population of marble butterflies was found on San Juan Island, Washington, "uplisting" this species from extinct to extirpated (Shepard 1999). Introduced plants and grazing by sheep in the early 1900s most likely eliminated or reduced the larval food plant. Shepard (1995) speculates that the introduction of the cabbage white (*Pieris rapae*) may also have had an impact on this regional endemic. Shepard (1999) suggests that undiscovered populations may occur on Sidney, James, and D'Arcy islands. Given the severe overgrazing by introduced fallow deer (*Cervus dama*) and early farm history on Sidney and James islands, and the lack of suitable habitat on D'Arcy Island, this seems unlikely. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) considered this species extirpated in Canada in 1999 (Shepard 1999).

Perdiccas Checkerspot (Euphydryas chalcedona perdiccas) G5T2T3 SX.— In Canada, this subspecies was formerly found on southern Vancouver Island (Layberry et al. 1998), where it was known from Goldstream, Mount Tzouhalem, and Duncan (Shepard 1995). It was last recorded in British Columbia in 1919 (Shepard 1995). Grazing by introduced sheep may have eliminated the larval food plant (Shepard 1995).

Viceroy (Butterfly) (Limenitis archippus) G5 SX.— Last collected in Lillooet in 1930 (Scudder 1996). This showy

butterfly is a well-known mimic of the monarch butterfly (*Danaus plexippus*) in both appearance and flight behaviour. Larvae of the species were recorded feeding on apple foliage in south-central British Columbia. Possibly extirpated due to pesticide use (Guppy et al. 1994). In other parts of the species range the larval food plants are usually willow (*Salix* spp.) and poplar (*Populus* spp.; Layberry et al. 1998), thus the loss of riparian habitats in south-central British Columbia (Cannings et al. 1987) may have also played a role in this extirpation.

Extirpated Reptiles

Western Pond Turtle (Clemmys marmorata) G3 SX.— Originally ranged from the lower Fraser Valley in southwestern British Columbia (Cannings et al. 1999) through to Baja California (Storm and Leonard 1995). Known in British Columbia from 2 records, both from south of the Fraser, and the last of which was in 1966. The species is now extirpated from the Puget Sound area of Washington State (Richardson 1993) and is listed as endangered in the remainder of Washington, with only 2 isolated populations remaining in the state (Richardson 1993). A recovery plan has been prepared for Washington State (Hays et al. 1999).

Gopher Snake, catenifer ssp. (Pituphis catenifer catenifer) G5T5 SX.— Recorded from Galiano Island and at the Sumas border crossing (Gregory and Campbell 1984), the last record was in the 1800s (Gregory and Gregory 1999). While no searches have been conducted for this subspecies, the species is large and resembles a rattlesnake, and it seems unlikely that it has been overlooked. The subspecies is considered extirpated in Washington State as well (Storm and Leonard 1995). Probably lost from habitat loss, direct persecution, and the impact of introduced species (Cannings et al. 1999).

Extirpated Birds

Sage Grouse (Centrocercus urophaianus) G5 SX.— Formerly widespread in western North America, but increasingly fragmented and declining (McAdam 1997). In British Columbia, sage grouse were restricted to the southern Okanagan, and the last record of the original population was a bird shot in 1918 (Campbell et al. 1990). A reintroduction of 57 sage grouse in 1958 failed, the last bird found from this reintroduction attempt being one found dead on Anarchist Mountain in 1966 (Campbell et al. 1990). The population in Washington State has undergone a dramatic decline and the species is no longer found in any county adjacent to British Columbia (Smith et al. 1997). Washington State's population is estimated at only 600 (Smith et al. 1997) to 800 (Stepniewski 1999) birds.

Yellow-billed Cuckoo (Coccyzus americanus) G5 SXB,SAN.— Most records are from the period between 1881 and 1927, and the species is presumed to have bred in southwestern British Columbia, including southern Vancouver Island (Campbell et al. 1990). Western populations may be a separate subspecies, *C. a. occidentalis* (Hughes 1999). There are a few British Columbia records since 1988, but the species is regarded as being extirpated as a breeding bird (Campbell et al. 1990, Fraser et al. 1999). This species has declined dramatically throughout the Pacific states (e.g., Laymon and Halterman 1987, Gaines and Laymon 1984, Hughes 1999). It was extirpated from Washington State by 1934, from Oregon by 1945, and from California north of Sacramento by the 1950s (Hughes 1999). The remaining western North America population is probably less than 30 pairs, all restricted to southern California (Laymon and Halterman 1989).

HISTORIC SPECIES

Historic Nonvascular Plants

The following nonvascular plants have not been verified in British Columbia in the past 45 years.

Bryum tenuisetum G? SH.— Listed as historic by the CDC. The only record in CDC files is of an 1887 collection from an unknown location in British Columbia.

Dicranum fuscescens var. congestum G5T?Q SH.— Known from 3 widely separated sites (Golden, Six Mile Lake, and the Skeena region), but last collected in 1931 (CDC data).

Polystricum commune var. perigoniale G?SH. — Last collected in 1926 at Summit Lake, New Denver.

Historic Vascular Plants

The following plants have not been verified in the past 45 years. For more detail on the rare vascular plants of British Columbia see Douglas et al. (1998); the brief descriptions that follow are from that volume. Plants listed as historic are Red-listed in British Columbia (Douglas et al. 1998). Plants are listed in alphabetical order by Latin name.

Giant-Hyssop (Agastache foeniculum) G4G5 SH.— Formerly found in open woods and clearings in the montane zone, this species was last collected in Golden in 1947 (Douglas et al. 1998). The species is available in the nursery trade, but no wild populations have been established from cultivated stock.

Alaskan Orache (Atriplex alaskensis) G3G4Q SH.— Known in British Columbia only from the Queen Charlotte Islands, and last collected in British Columbia in 1898 (Douglas et al. 1998).

Short-Flowered Evening-Primrose (Camissonia breviflora [= Oenothera brevifolia]) G5 SH.— Last collected in British Columbia in 1925 in dry steppe near the Neamaia River (Douglas et al. 1998).

Boreal Paintbrush (Castilleja fulva) G1? SH.— A poorly known species apparently restricted to northeastern British Columbia and southwestern Yukon. It was last collected in British Columbia in 1940 (Douglas et al. 1998).

Crawe's Sedge (Carex crawei) G5 SH.— Formerly

collected in southeastern British Columbia. Last collected in 1948 at Kinbasket. Douglas et al. (1998) report that this site may no longer exist.

Brookgrass or Water Hairgrass (Catabrosa aquatica) G5 SH.— Known only from a single collection from Fernie in 1911 (Douglas et al. 1998).

Golden Carpet (Chrysosplenium iowense) G3 SH.— In British Columbia known only from the Tupper Creek area near Dawson Creek. Last collected in 1937 (Douglas et al. 1998).

Fendler's Cryptantha (Cryptantha fendleri) G4 SH.— Formerly found in dry sites in the steppe and montane zones of the Douglas-fir and ponderosa pine zone in south-central British Columbia. Last collected in 1937 (Douglas et al. 1998).

Smooth Spike-Primrose (Epilobium pygmaeum) G5 SH.— Last collected in 1921 from a "Douglas Lake," exact locality unknown, as there are several Douglas Lakes in British Columbia (Douglas et al. 1998).

Globe Gilia (Gilia capitata var. capitata) G5T5 SH.— Original populations were found on dry sites in the Interior of British Columbia. Recent collections from Victoria are regarded as escaped cultivated plants (Douglas et al. 1998).

Shy Gilia (Gilia sinuata) G5 SH.— Known only from a collection at Osoyoos in 1940 (Douglas et al. 1998).

Floating Water Pennywort (Hydrocotyle ranunculoides) G5 SH.— Last collected in Departure Bay in 1939. Formerly known in British Columbia from southeastern Vancouver Island and the lower Fraser Valley (Douglas et al. 1998). Probably extirpated, as the ponds, marshes, and wet sites that it was known from are now urban areas (Douglas et al. 1998 including addenda).

Arctic Daisy (Leucanthemum arcticum) G4? SH.— In British Columbia known only from an 1893 collection on Larcomb Island in Observatory Inlet, on the extreme northern British Columbia coast (Douglas et al. 1998).

Spurred Lupine (Lupinus arbustus ssp. neolaxiflorus) G5T? SH.— In British Columbia known only from a 1944 collection at Flagstone, in extreme southeastern British Columbia (Douglas et al. 1998).

Sulphur Lupine (Lupinus oreganus var. kincaidii [Lupinus sulpureus ssp. kincaidii]) G5T2 SH.— Known in British Columbia from a collection made in Victoria in 1929 (Douglas et al. 1998).

Branching Montia (Montia diffusa) G4 SH.— In British Columbia known only from a collection in 1916 in moist forest near Port Alberni, Vancouver Island (Douglas et al. 1998).

Pine Broomrape (Orobanche pinorum) G4 SH.— In British Columbia known only from a collection made in 1914 at Duncan along the Cowichan River (Ceska 1986, Douglas et al. 1998).

Canada Ryegrass (Oryzopsis canadensis) G5 SH.— Known in British Columbia from a single species collected at Pouce Coupe in 1921 (Douglas et al. 1998).

Northern Parrya (Parrya nudicaulis) G5 SH.— In British Columbia last collected in 1945 from Log Jam Creek and Father Mountain near the British Columbia–Yukon border (Douglas et al. 1998).

Fringed Pinesap (Pleuricospora fimbriolata) G4 SH.— In British Columbia known only from Horne Lake on Vancouver Island. Last collected in 1916 (Douglas et al. 1998).

Banff Bluegrass (Poa laxa ssp. banffiana) G5?T1 SH.— Known in British Columbia only from the Rocky Mountains at Simpson Pass and the Valley of Ten Peaks. Last collected in 1943 (Douglas et al. 1998).

Coastal Bluegrass (Poa nervosa) G5 SH.— First collected by Scouler at Nootka Sound and named by Hooker in 1840 based on the British Columbia specimen. The species has not been collected from British Columbia since then (Douglas et al. 1998).

Virginia Polypody (Polypodium virginianum) G5 SH.— Known in British Columbia from Beatton River and Mount Selwyn in northeastern British Columbia. Last collected in 1943 (Douglas et al. 1998).

California Sword-Fern (Polystichum californicum) G4 SH.— Known in Canada only from Texada Island, southwestern British Columbia, where it was last collected in 1937 (Douglas et al. 1998).

Purple Rattlesnake-Root (Prenanthes racemosa ssp. multiflora) G5T4? SH.— Formerly known from Dawson Creek and Pouce Coupe in northeastern British Columbia. Last collected in 1946 (Douglas et al. 1998).

Siberian Primrose (Primula nutans) G5 SH.— Known in British Columbia only from Atlin, northwestern British Columbia, where it was last collected in 1914 (Douglas et al. 1998).

Prairie Buttercup (Ranunculus rhomboideus) G4 SH.— Formerly found in northeastern British Columbia in dry grasslands and open forests. Last collected at Hudson's Hope in 1934 (Douglas et al. 1998).

Alkali-Marsh Butterweed (Senecio hydrophilus) G5 SH.— A species of wet areas, often alkaline, formerly known from the Salmo and Kootenay river areas in extreme southeastern British Columbia. There no collections from British Columbia since 1929 (Douglas et al. 1998).

White Western Groundsel (Senecio integerrimus var. ochroleucus) G5T? SH.— Last collected in south-central British Columbia "prior to 1900" (Douglas et al. 1998).

Munroe's Globe-Mallow (Sphaeralcea munroana) G4 SH.— Last collected in British Columbia in 1922. Formerly found near Osoyoos (Douglas et al. 1998).

Salt-Water Cress (Thellungiella salsuginea) G4G5 SH.— Formerly found along dry, saline lakes and meadows at Windermere and Columbia Lake in southeastern British Columbia. Last collection was in 1942 (Douglas et al. 1998).

Historic Molluscs

Three-ridges Valvata (*Valvata tricarnata*). Last collected at Kootenay Lake in 1969 (J. Lee, pers.comm).

Historic Insects

Scudder (1996) lists a number of insects that have not been recorded for many years. Several of these have only been collected from British Columbia and, until they are found elsewhere, are considered endemic. However, many of these species have not been recently searched for and would require specialists to identify them. The CDC lists 2 species as historic, both from well-known and showy insect groups; one is a tiger beetle, the other a butterfly.

Parowana Tiger Beetle (Cicindela parowana) G4 SH.— A beetle of alkaline flats, known historically from Okanagan Falls, Oliver, and Penticton. There are no recent records and the Penticton site has been destroyed by a housing development (Scudder 1996).

Vancouver Island Blue (Plebejus saepiolus insulanus) G5TU SH.— No extant populations of this endemic Vancouver Island subspecies are known and it may be extinct (Guppy et al. 1994). Elsewhere the larvae feed on legumes, including both native and introduced clovers (Scott 1986), which are abundant on Vancouver Island (Shepard 1995). Last records for the species were from G. A. Hardy in 1963 at Francis Park and Observatory Hill, Saanich; and in 1960 at Mount Finlayson, now part of Goldstream Provincial Park, at Spectacle Lake on the Malahat, and at another unspecified Malahat (Station?) location (Shepard 1995).

Historic Reptiles

Pigmy Short-horned Lizard (Phrynosoma douglasi) G5 SH.— Known in British Columbia from a collection of 2 individuals near Osoyoos in 1910 (Gregory and Campbell 1984, Powell and Russell 1992), as well as an unconfirmed record from Chopaka in the 1980s and several unconfirmed sight records from southeast of Oliver in the past 30 years (Cannings et al. 1999). Extensive habitat loss to orchards, urban areas, and vineyards makes extirpation likely.

Historic Birds

Horned Lark, strigata ssp. (Eremophila alpestris strigata) G5T2 SH.— This coastal subspecies of horned lark formerly occurred in British Columbia in open habitats on southeastern Vancouver Island and in the lower Fraser Valley (Campbell et al. 1997). The last summer coastal records of the horned lark were at the Vancouver International Airport in 1987 (Campbell et al. 1997). The species probably disappeared due to urbanization of habitat and habitat structural changes caused by the introduction of species such as Scotch broom, gorse, and tall agronomic grasses (Fraser et al. 1999). This subspecies is also extirpated from most of its former Washington range (Smith et al. 1997). The estimated Washington State population in 1999 was "fewer than 100 pairs; this is down from perhaps thousands from the turn of the century" (R. Rogers, Evergreen State College, pers. comm.).

Historic Mammals

Southern Red-Backed Vole, occidentalis ssp. (Clethrionomys gapperi occidentalis) G5T5 SH.— Formerly considered a separate species (*C. occidentalis*; Cowan and Guiguet 1965). Known in British Columbia only from 2 specimens, 1 from Point Gray and another from Stanley Park. Not recorded for more than 40 years (Cannings et al. 1999).

Long-Tailed Weasel, altifrontalis ssp. (Mustela frenata altifrontalis) G5T? SH.— Known from the Fraser Delta north to Harrison Lake (Forbes n.d., Cowan and Guiget 1965). No recent specimen records, and only 1 recent, unconfirmed sight record exists (Cannings et al. 1999). The subspecies is apparently relatively abundant in second-growth forests in nearby Washington State (Wilson and Carey 1996).

Sei Whale (Balaenoptera borealis) G3 SH.— Sei whales were formerly encountered in British Columbia waters between June and August (Horwood 1987). More than 1,400 sightings were made of sei whales by Coal Harbour whalers off Vancouver Island and the Queen Charlotte Islands between 1963 and 1967 (Cirella 1999). A total of 3,272 sei whales were taken in British Columbia waters, 62% between 1962 and 1967. However, Cirella (1999) could find no recent sightings of the species since the commercial whaling era on the west coast.

Black Right Whale (Eubalaena glacialis) G1 SH.— This was the first whale species to be decimated by commercial whaling in British Columbia. It was considered commercially extinct by the mid-1800s, although there may have been illegal takes of right whales in the North Pacific into the 1960s (Brownell and Clapham 1996). The surviving population in the North Pacific is estimated to be in the low hundreds (Gaskin 1990). The few remaining animals appear to congregate in the summer months around the eastern Aleutian and Kodiak islands and the nearby waters of the south Bering Sea. Wintering grounds are still unknown. Last recorded black right whale for British Columbia was one taken accidentally off Vancouver Island in 1951 (Cirella 1999). However, a sighting on 24 May 1992 off Cape Elizabeth, Washington (Rowlett et al. 1994) and a small number of sightings in Oregon, California, and Baja California (Brownell and Clapham 1996) indicate that very small numbers may occasionally pass through British Columbia waters.

EXTIRPATED SPECIES SUCCESSFULLY REINTRODUCED

In addition to the species listed below, reintroduction attempts have been made for sage grouse, but the population slowly declined and eventually disappeared (Campbell et al. 1990). Burrowing owls, while never regarded as officially extirpated (Fraser et al. 1999), are dependent on an ongoing reintroduction program (Leupin and Low 1999).

Sea Otter (Enhydra lutra) G4 S2.— This species formerly inhabited coastal regions of the north Pacific, but by 1911

it had been extirpated from British Columbia and most of its eastern Pacific range, with only peripheral populations remaining (Watson et al. 1997). A total of 89 animals were reintroduced into British Columbia between 1969 and 1972 (Watson et al. 1997). Sea otter numbers are estimated to be 2,000–2,200 on the west coast of Vancouver Island and 200–400 at the Hakai Provincial Recreation Area (Cannings et al. 1999).

Plains Bison (Bison bison bison) G4 S3.— Plains bison formerly occurred in very small numbers in the extreme eastern part of the province (Cowan and Guiget 1965). A number of archaeological sites in southeastern British Columbia contain bison bones, however, and clearly the species was more widespread than the few records would indicate. The only extant population is centred around Pink Mountain in northeastern British Columbia and is the result of establishment of feral animals. It is outside the historic range of this subspecies (Cannings et al. 1999). In 1998 the British Columbia population was estimated to be between 1,500 and 1,800 animals (Cannings et al. 1999).

Wood Bison (Bison bison athabascae) G4T? S1.— Historically, wood bison were found in northeastern British Columbia, and prehistorically near Atlin as well (Wood Bison Recovery Team 1987). They were extensively harvested in the 1800s (Harper et al. 1999) and the last confirmed record was a single animal shot in 1906 near Fort St. John (MacGregor 1952). A male was reported as shot by natives in the Liard River drainage near Lower Post in 1939 (Clarke 1944, Lotenberg 1996). In the 1960s there were sightings in the Clear Hills north of Fort St. John on the British Columbia–Alberta border, and in the late 1970s a small herd was observed in the lower Buckinghorse River area (Harper et al. 1999). Wood bison reestablished populations in the 1980s from animals dispersing from the Nahanni Butte reintroduction in the Northwest Territories. Additional reintroductions were made in the province in 1995, 1996, and 1999 (Harper et al. 1999). There are now 80–100 animals in the herds found in British Columbia and another 100 animals in the Hay-Zama herd that is shared with Alberta (Harper et al. 1999).

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Arthropod Biodiversity of Northern Temperate Ancient Rainforests: Going, Going...Gone!

N.N. Winchester

Biology Department, University of Victoria, PO Box 3020, Victoria, BC, V8W 3N5, Canada
tundrast@uvvm.uvic.ca

ABSTRACT

The biodiversity crisis in ancient northern temperate rainforests continues to be accelerated by rapid habitat loss and consequent extinctions of floral and faunal species assemblages that cannot adjust to rapid, and often large scale, habitat alterations. In an effort to record arthropod biological diversity in these forests I have studied arthropods in a number of Vancouver Island rainforest types since 1992. Based on these findings I address the following questions: (1) Do ancient forest canopies contain an endemic and largely undescribed arthropod fauna? (2) Do ancient forest ground dwelling arthropod species exist in second growth forests? (3) Can forest silvicultural practices recreate the habitat needed to maintain the full complement of ancient forest arthropod species? Answers to these questions are needed to address the issues that surround the maintenance of biological diversity (form and function) in these ancient forests.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

How Do Amphibians Go Extinct?

David M. Green
Redpath Museum, McGill University
859 Sherbrooke Street West, Montreal, PQ, H3A 2K6, Canada
davidg@shared1.lan.mcgill.ca

ABSTRACT

Many causes have been proposed to explain the global decline of amphibian populations. These include acid precipitation, UV-b radiation, epidemic diseases, deformities, habitat disruption, etc. Amphibians are noted for their bi-phasic life histories, supposedly limited abilities to disperse, and permeable skins, yet amphibians also exhibit enormous variation in life histories and can maintain large populations. How do these intrinsic factors influence expectations of decline? Two sorts of declines are recognizable: a downward trend in numbers of individuals within populations, and a downward trend in numbers of populations. Populations decline in size when the numbers of new recruits and immigrants do not keep pace with losses of individuals due to death and emigration. Such a trend may be impossible to demonstrate. In similar fashion, though, the number of populations of a species is governed by the relative rate of losses of existing populations (local extinctions) versus colonization of new populations. This is more germane. For the total number of populations to decrease (i.e., range contraction or species decline) either the rate of local extinction increases or the rate of recolonization decreases, or both. Parameters influencing population persistence include environmental stochasticity, demographic stochasticity, catastrophes, density dependence, intrinsic rate of increase (r), the rescue effect of metapopulations, and the storage effect of overlapping generations. An 11-year demographic study of Fowler's toads (*Bufo fowleri*) at Long Point, Ontario illustrates stochastic and deterministic factors influencing severe population size fluctuation. This is typical of pond-breeding amphibians, which have high maximum r, very large clutch sizes, very high mortality rates, low effective population sizes compared to census population sizes, fluctuations, and propensity to local extinctions. Contrarily, plethodontid salamanders have stable population sizes. Characteristic for such terrestrial-breeding amphibians are low maximum r, very small clutch sizes, very low mortality rates, high effective population sizes, and persistent populations. The 2 different amphibian demographics predict differing extinction probabilities in the face of the 3 evils of habitat destruction, habitat disruption, and habitat degradation. Those amphibians in decline in Canada tend to be pond-breeders, which we should expect to be most severely affected by disruption of habitats despite, or because of, their high clutch sizes and reliance upon dispersal.

Key words: amphibian, *Bufo fowleri*, extinction, Fowler's toad, *Plethodon*, salamander.

Gathering together in 1989 at the First World Congress of Herpetology in Canterbury, England, herpetologists swapping stories over beer and crisps first came to a realization that frogs might be in decline worldwide. One story after another of frogs and toads disappearing, getting rarer, or simply not being where they used to be raised no small amount of worry among the participants (Wake 1991). The following year, 1990, in Irvine, California, a hastily convened conference met to ask if there was indeed evidence of extinctions among amphibian populations. Were there genuine losses greater than background extinction rates? Were amphibians bio-indicators of greater ecological disaster? In this climate of crisis, the Declining Amphibian Populations Task Force (DAPTF) was formed under the Species Survival Commission (SSC) of the World Conservation Union (IUCN) in 1991 (Vial and Saylor 1993). That same year, a Canadian working group assembled and called itself Declining Amphibian Populations Canada (DAPCAN; Green 1997*a*).

If amphibians were in decline, what was the cause? Was there some global plague killing them off? Contaminants were implicated: acid precipitation and toxic chemicals can kill eggs and embryos (Berrill et al. 1997, Grant and Licht 1997). UV-b radiation was fingered: there is evidence that some species have greater repair abilities (the enzyme photolyase) than others (Blaustein et al. 1994*a*). Were there epidemic diseases caused by chytrid fungus; "redleg" bacteria infections due to *Aeromonas*, *Pseudomonas*, or other pathogens; or viral diseases such as *Iridovirus* (Cunningham et al. 1996, Lips 1999)? Lately, amphibian

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

deformities have been discussed: are they due to trematode infections disrupting development, or to environmental contaminants inducing "birth defects" (Sessions and Ruth 1990, Bonin et al. 1997, Ouellet et al. 1997)? And finally, habitat change leading to isolation of populations, interference with dispersal, and destruction of metapopulation structure was suspected surely to be a cause for decline (Bradford et al. 1993, Blaustein et al. 1994*b*, Dupuis 1997).

WHAT IS A DECLINE?

Vial and Saylor (1993) defined decline as "... a definite downward trend in numbers over a span of time appropriate to the species' life history, shown to be in excess of the normal fluctuations in population size." This definition specifies studying demography and numbers of animals. One way to describe change in population size is a familiar finite difference equation from population biology:

$$N_t = N_{t-1} + (B + I) - (D + E).$$

This equation simply says that the number of animals (N) in the population at some time (t) depends on the number that were there before (t - 1), plus new ones added by birth (B) and immigration (I), minus those lost to death (D) and emigration (E). Thus it is a simple birth/death model balancing gains (G) versus losses (L; Poethke et al. 1996). The rate of increase (r) is G - L. The magnitude of these parameters, as well, is important for understanding the demography of a population. When the magnitudes of gains and losses are high, there is high turnover of individuals. Small percent changes in parameters translate into large numbers of individuals lost or gained. Where losses exceed gains, r is negative and there is decline.

The birth/death process model works for numbers of populations, too. Populations arise by founders arriving from other, pre-existing populations. Populations also may die out; that is, they go locally extinct. Where losses exceed gains, there is decline in the range of the species. Where gains exceed losses, the range expands (Fig. 1).

There are, therefore, 2 types of decline. Decline within populations is about how populations get small and/or how long populations persist; these are the 2 conservation paradigms discussed by Caughley (1994). The other sort of decline is decline across populations, and it concerns how ranges get small or how long species persist. I contended that this aspect is more germane to the problem of declining amphibians than the demographic approach, although demography is still of key importance (Green 1997*a*). I defined decline in this context as: "The condition whereby the local loss of populations across the normal range of a species so exceeds the rate at which populations may be established or reestablished that there is a definite downward trend in population number" (Green 1997*a*).

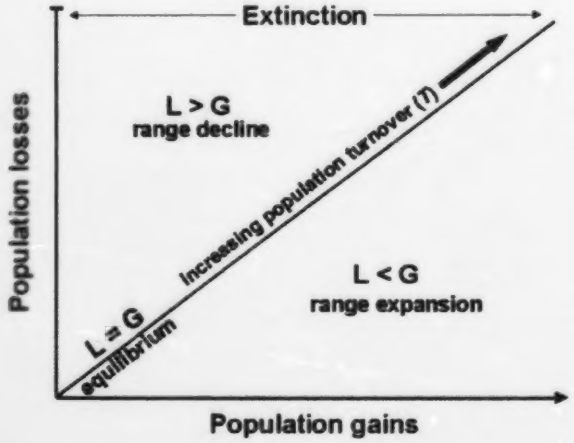

Figure 1. Relationship of population losses to population gains in relation to extinction risk and turnover rate. Declines occur whenever losses exceed gains, resulting in contraction of range. Expansion occurs whenever gains exceed losses, resulting in increased range. As the magnitude of losses or gains increases, risk of extinction through stochastic forces increases, and total population loss may result, which is unrecoverable.

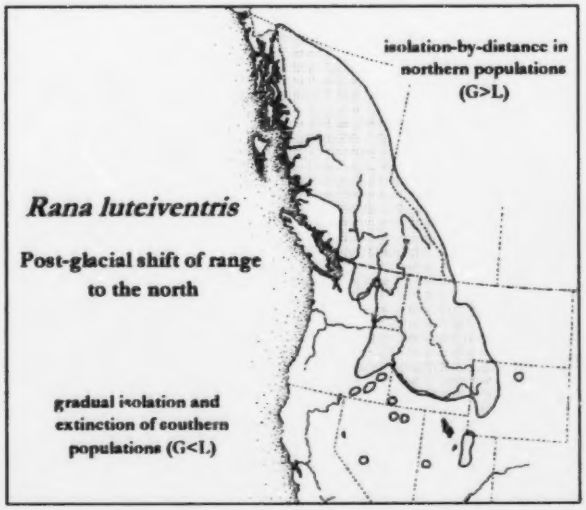
isolation-by-distance in northern populations (G>L)
Rana luteiventris
Post-glacial shift of range to the north
gradual isolation and extinction of southern populations (G<L)

Figure 2. The range of the Columbia spotted frog (after Green et al. 1997) illustrates a shift in range over post-glacial times resulting in a characteristic fragmentation of the range in the south, where losses of populations have exceeded gains.

EXPANSION VERSUS CONTRACTION

As is clear from the birth/death models, decline is the flip-side of increase. Both gains and losses must be considered to understand either. Clearly, to decrease total number of populations there may either be an increase in the rate of local extinctions (raised L) or a decrease in the rate of recolonization (lowered G; Fig. 1).

A natural example of this principle is afforded by the Columbia spotted frog, *Rana luteiventris* (Green et al. 1996). The post-glacial shift of this species' range to the north (Fig. 2) resulted in gradual isolation and extinction of southern populations, where population losses exceeded gains ($L > G$), and isolation-by-distance in northern populations, where gains exceeded losses ($G > L$). In both expanding and contracting range margins there is reduced genetic variability, at the expanding edge due to founder effects, and at the contracting edge due to bottlenecks and inbreeding. Thus higher population turnover (greater G or greater L) leads to overall lower genetic effective population sizes.

FLUCTUATING POPULATIONS

The parameters influencing population persistence are well known (Lande 1993) and include elements of chance (demographic stochasticity, environmental stochasticity, catastrophes) and determinism (density dependence, intrinsic rate of increase [r], rescue effect in metapopulations, storage effect of overlapping generations). An example will illustrate these interacting forces.

I have conducted 11 years of study of Fowler's toads (*Bufo fowleri*) at Long Point, Ontario, a 35-km sand spit on Lake Erie (Green 1997*b*, Green unpubl. data). It is the northern limit of Fowler's toads and an extremely dynamic environment. Severe storms in 1985–86 demolished the shoreline, as well as the toad population, allowing the cycle of population recovery to be studied in detail. I use an intensive approach to determining population size. Every night, without fail, during the toads' breeding season, my students and I have gone out and captured every possible adult toad, marked them all, and released them. Our capture and recapture rates are very good (Fig. 3) enabling us to very closely estimate the true population size to within 5–10% using a removal sampling protocol (Chao et al. 1992). The population fluctuates widely in size (Fig. 4) and in age structure (Fig. 5), and has gone through successive phases of immigration and high recruitment, high winter-kill from an unusually cold winter in 1994 (a stochastic event), then low recruitment as ecological succession removed former breeding sites from production. The toads prefer clear water and sandy bottomed ponds.

The result has been a fluctuating population size for Fowler's toads at Long Point. This is, in fact, normal for many amphibian populations (Pechmann et al. 1991, Berven 1995). Where populations fluctuate, there is usually insufficient power in statistical tests to detect true declines in numbers. The ad hoc causes defy generalizations. The high variance in r and N is characteristic of pond-breeding amphibians, with their very large clutch sizes (about 2,300 eggs in *B. fowleri*), very high mortality rates, high r_{max}, low effective population sizes compared to census population

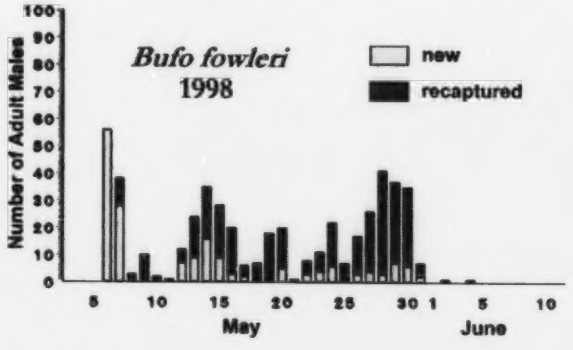

Figure 3. Success rate for nightly capture and recapture of male Fowler's toads at Long Point, Ontario during and after the breeding season, 1998. Toads emerged on 6 May. After 31 May, breeding was effectively over. Comparatively few new individuals were caught after mid-May, indicating a high proportion of the population had been sampled, yielding an accurate estimate of population size.

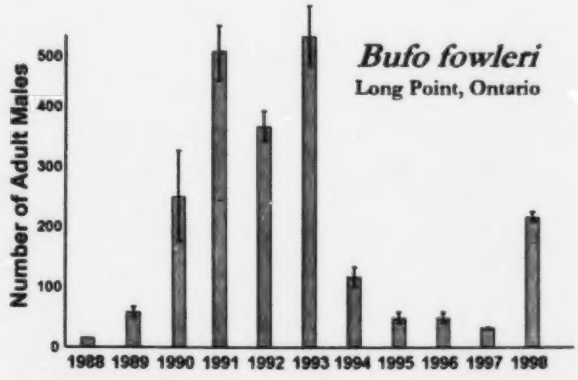

Figure 4. Population size estimates for male Fowler's toads at Long Point, Ontario, based on the sort of data depicted in Figure 3. Bars are 95% confidence limits. The population rose and fell more than an order of magnitude in size due to both deterministic and apparently stochastic influences.

Table 1. Examples of amphibian demographics.

Species	Approximate mean clutch size	Survival rate (egg to adult)	Maximum age (yr)	Reference
Plethodon jordani	15	0.119	25	Hairston 1987
Triturus vulgaris	200	0.0015	13	Bell 1977
Rana temporaria	1,300	0.0012	8	Riis 1991
Bufo fowleri	2,300	0.0007	5	Green unpubl. data

size, and dependence upon dispersal. They should be expected to turn over at a high rate and be subject to "Gambler's Ruin" (Caughley 1994; Fig. 6). Even when the odds are even, gamblers eventually go bust simply because of a basic asymmetry. There is no upper limit to winnings from which a gambler cannot always lose again, but there is definitely a lower limit, zero, from which a gambler cannot recover. The higher the stakes (turnover in population terms), the greater the risk.

STABLE POPULATIONS

But not all amphibian populations behave like pond-breeding species with aquatic larvae like *B. fowleri* (Table 1). Terrestrial direct-developing (with a larva) plethodontid salamanders such as *Plethodon glutinosus* may have great population stability (Fig. 7), and populations of many other amphibians without larvae are regulated (Hairston 1987). Terrestrial-breeding species have very small clutch sizes, very low mortality rates, low r_{max}, high effective population sizes compared to census population sizes, and low rates of dispersal (Petranka 1998). The low population turnover and low variance in r and N characteristic of terrestrial-breeding amphibians makes it possible to detect population size declines statistically. These stable populations persist, even at lower population sizes, because they operate at lower levels of turnover (Fig. 6).

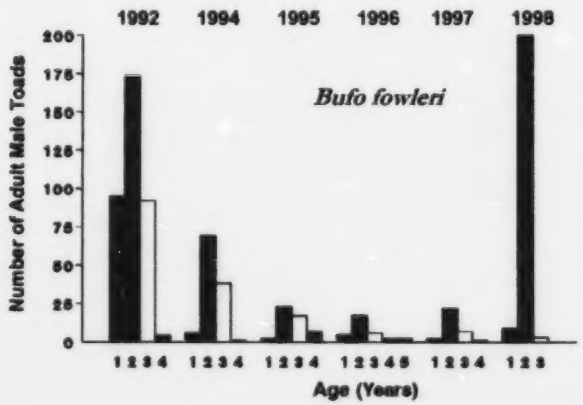

Figure 5. Changing age structure of adult male Fowler's toads in 6 years at Long Point, Ontario. Aside from changes in number of individuals, there are large changes in proportions of adults aged 1–4 years, or, rarely, 5 years old.

Table 2. Two amphibian demographics.

High-risk Pattern	Low-risk Pattern
high fecundity	low fecundity
high mortality	low mortality
high dispersal ability	low dispersal ability
no territoriality	territoriality
leads to:	leads to
high stochasticity fluctuations	low stochasticity stability
e.g., *Bufo*	e.g., *Plethodon*

DIFFERENTIAL RESPONSE TO THREE EVIL TRIPLETS

There are clearly 2 extreme demographics among amphibians, which can be exemplified by *Plethodon* and by *Bufo* (Table 2). A species with a high intrinsic turnover rate, both of individuals and of populations, constantly flirts with local extinction (Fig. 8). It compensates with high dispersal (Sinsch 1997). Terrestrial breeding, direct-developing species may have very low dispersal rates (Petranka 1998), unlike pond-breeding species (Breden 1988).

What we must now ask is how these different kinds of amphibians, with different demographic characteristics and levels of individual and population turnover, react when the environment is perturbed. How do they go extinct? There are 3 categories of perturbation:

- Habitat Destruction, which is the outright loss of necessary habitats.
- Habitat Fragmentation, which is the isolation of habitat components curtailing dispersal.
- Habitat Degradation, which is any alteration of habitats by various means (contaminants, alien predators, introduced diseases, etc.), rendering it less hospitable than it used to be.

Different amphibians should react to these 3 "Evil Triplets" in characteristic ways (Table 3). Most significantly, those species that have high turnover rates and high r_{max},

Table 3. Expected responses of local amphibian populations to environmental impacts.

Effect	Expected Population Level Response	
	High r_{max} species[a]	Low r_{max} species[a]
Habitat destruction	Local extinction.	Local extinction.
Habitat fragmentation	Local stochastic extinctions remain permanent. Eventual losses of populations over extended areas.	No necessarily significant effect.
Habitat degradation	Variable negative effects. Local extinctions, but may recover (rescue effect).	Local extinctions unlikely but permanent (negligible rescue effect)

[a] r_{max} = maximum intrinsic rate of increase.

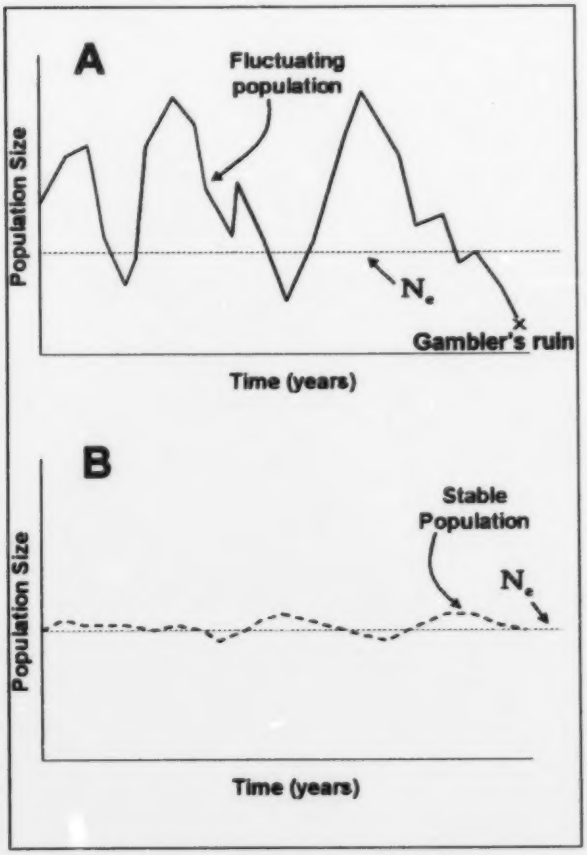

Figure 6. Fluctuating populations have effective population sizes Ne (calculated as the harmonic mean size) that are lower than the arithmetic mean, and risk extinction via "Gambler's Ruin." Stable populations have Ne close to the arithmetic mean, and have lower risk of stochastic extinction.

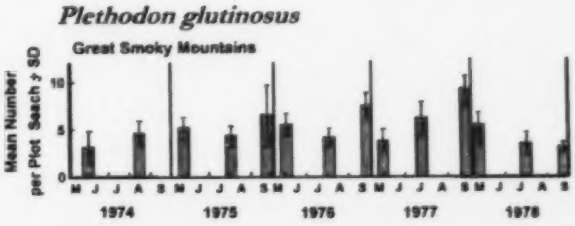

Figure 7. Stable population size in *Plethodon glutinosus* (after Hairston 1987). The density of salamanders rises each fall as eggs hatch, and declines each spring.

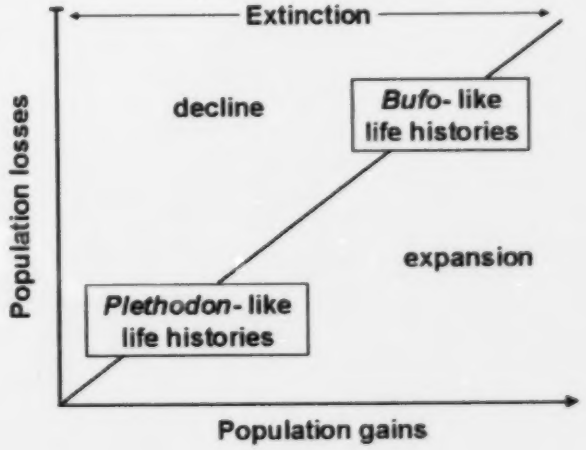

Figure 8. High turnover species with Bufo-like life histories risk stochastic extinction. Populations can be rescued by dispersal.

and which require dispersal for recolonization of sites after inevitable local extinctions, should be profoundly more affected by habitat fragmentation than species with low turnover rates. Therefore, the species that may be most at risk are not necessarily those with low reproductive rates. Species with high variance in r and N, with complex habitat requirements, low effective population sizes, and dependence upon dispersal, (i.e., pond-breeders among amphibians), might be the ones most at risk, despite their high fecundities. Of those species of amphibians suspected to be in decline in Canada, almost all are pond-breeders (Weller and Green 1997).

ACKNOWLEDGEMENTS

This work is supported by a Natural Sciences and Engineering Research Council of Canada research grant. My studies of Fowler's toads are supported by an Endangered Species Recovery Fund Grant from World Wildlife Fund Canada and the Canadian Wildlife Service. I also thank the many students who have assisted me over the years in the field and in the lab.

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Endangered Species Protection in Alberta: "Where's the Beef?"

Wendy Francis[1]
Conservation Director, Calgary/Banff Chapter
Canadian Parks and Wilderness Society
306-319 Tenth Avenue S.W., Calgary, AB, T2R 0A5, Canada

ABSTRACT

The rapid pace of species extinctions globally has been called one of the most pressing environmental issues facing humanity. Canada has made national and international commitments to protect biodiversity, particularly endangered species. Canada's response to these commitments is complicated by a divided constitutional jurisdiction over the environment. The federal, provincial, and territorial governments have endeavoured to find a solution to this shared responsibility by executing the National Accord for the Protection of Species at Risk (National Accord.) In it, each jurisdiction commits to take specific steps to implement legal protection for endangered species. Alberta's answer to endangered species protection has been to enact amendments to the Wildlife Act. Although passed before the National Accord was signed, these amendments are all Alberta intends to do to fulfil its responsibilities. This paper will analyse those provisions and the extent to which they meet Alberta's commitments under the National Accord, using specific examples of species and habitats. It concludes that the broad discretionary powers given to Alberta's Minister of Environmental Protection and the nature of the process provide little guarantee of protection for endangered species.

Key words: Alberta, legislation, National Accord.

Many conservation biologists agree that the Earth is facing a mass extinction episode. It threatens to be as comprehensive as the one about 65 million years ago that wiped out not only the dinosaurs, but 76% of all species. The average rate of extinction is only about 1 species in any major group every million years. Some scientists believe the planet's transformation could see the extinction of 50–66% of all species before it is over (Quammen 1998).

Although other parts of the world, particularly those with rain forests, are the focus of most popular concern, Canada also is faced with an increasing number of species at risk. Currently, about 300 Canadian species are identified as being at risk of extinction by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC).

The causes of the current extinction crisis are well known. Extensive habitat loss to agriculture, urban development, and resource extraction is the primary cause of species loss. Contributing factors are habitat fragmentation, overhunting and overfishing, invasive species, and the secondary consequences of other extinctions (Quammen 1998). In other words, the efforts of just 1 species to satisfy its growing population's demands for goods, services, and activities are appropriating most of the Earth's resources and threaten the very existence of the balance of the planet's creatures.

The Canadian government and the provinces have made some noble commitments to take action to halt or reverse the trend toward species extinction. Canada was one of the first signatories to the global Convention on Biological Diversity. This document requires countries to take various steps to identify and preserve threatened species, habitats, and ecosystems.

In Canada, the effort to reduce factors leading to species extinctions is complicated by a divided constitutional jurisdiction. Provincial governments have exclusive jurisdiction to manage "resources" within their boundaries, which traditionally has been interpreted to include wildlife resources. The constitution gives the federal government exclusive jurisdiction over migratory birds and fisheries. The federal government has no express power to legislate regarding the environment or other resources.

However, the Supreme Court of Canada has determined that the environment is such a broad subject area that both federal and provincial levels of government can pass laws for environmental protection (Oldman Dam case, Supreme Court of Canada; Canada 1992). Further, the federal government has a residual power to make laws for the "peace, order, and good government of Canada." Many scholars have interpreted this to give the federal government authority to

[1] Present address: 12 Grozier Road, Cambridge, MA 02138, USA
wendyfrancis@compuserve.com

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

regulate national emergencies or issues possessing a "national dimension," such as species in danger of extinction (Gibson 1996).

Even though the federal government appears to have sufficient constitutional power to enact a law protecting endangered species everywhere in Canada, political sensitivities about federal/provincial relations have made its approach to endangered species protection timid. At a time when a strong federal presence would antagonize Quebec, other provinces have asserted their own right to make endangered species laws within their boundaries.

The federal, provincial, and territorial governments have tried to find a solution to the mutually beneficial premise of shared responsibility by executing the National Accord for the Protection of Species at Risk (National Accord). In it, each jurisdiction commits to take specific steps to implement legal protection for endangered species.

Alberta's answer to endangered species protection has been to enact amendments to the Wildlife Act. Although passed before the National Accord was signed, these amendments are all Alberta intends to do to fulfil its responsibilities (Alberta Environmental Protection staff, 1996, pers. comm.). This paper will analyze those provisions and the extent to which they meet Alberta's commitments under the National Accord. It concludes that the broad discretionary powers given to Alberta's Minister of Environmental Protection and the process itself provide little protection for endangered species. Alberta's efforts fail to fulfil its responsibilities under the National Accord.

NATIONAL ACCORD FOR THE PROTECTION OF SPECIES AT RISK

The National Accord was signed by the federal, provincial, and territorial governments on September 27, 1996. In accordance with its provisions, each level of government has committed to, among other things, establish complementary legislation and programs that provide effective protection of species at risk and:

- provide an independent process for assessing the status of species at risk;
- legally designate species as threatened or endangered;
- provide immediate legal protection for threatened or endangered species;
- provide protection for the habitat of threatened or endangered species;
- provide for the development of recovery plans within 1 year for endangered species and 2 years for threatened species; and
- consider the needs of species at risk as part of environmental assessment processes.

Since the execution of the Accord, the federal and provincial governments have struggled with a legislative formula that will both meet their requirements while preserving each jurisdiction's sense of constitutional propriety. The federal government's effort, Bill C-65, was both too weak to satisfy conservationists and too prescriptive for many provincial governments, landowners, and resource managers. It expired when Parliament dissolved in June 1997. Since then, consultations have revealed even less of an appetite among private property, resource, and agricultural constituents for prescriptive federal legislation. Another version of a federal bill, anticipated to be even weaker than C-65, is expected in the spring of 1999.

Table 1. Species on the Red List in Alberta

swift fox
burrowing owl
piping plover
Canadian toad
northern leopard frog
wood bison
peregrine falcon
whooping crane
great plains toad

ENDANGERED SPECIES PROTECTION IN ALBERTA

Alberta does not have "stand alone" legislation to protect endangered species. Rather, endangered species currently are managed in 2 ways: (1) the nonbinding list, *The Status of Alberta Wildlife*, and (2) endangered species provisions in Alberta's Wildlife Act.

The *Status of Alberta Wildlife* list provides information only; it has no regulatory protection for species at risk (Alberta Ministry of Environmental Protection 1992). It also assesses only mammals, birds, amphibians, and reptiles; the status of fish, plants, and insects in Alberta is not officially reviewed. Under the latest *Status of Alberta Wildlife* list, 29 species are at risk in Alberta: 9 species are Red-listed (they are at risk of dying out in Alberta; Table 1), and 20 animals are Blue-listed (they may be at risk of dying out in Alberta; Table 2).

ALBERTA'S WILDLIFE ACT

Alberta's regulatory efforts to preserve endangered species are found in its Wildlife Act (Alberta 1984). The statute primarily is directed toward managing wildlife populations. It classifies wildlife as game and nongame species, and contains the regulations for hunting, trapping, and possessing wildlife.

The Wildlife Act also contains limited provisions regarding the listing of and protection for endangered animals (but not plants or other life). Eleven animals are listed under the Wildlife Act. The swift fox, plains bison, whooping crane, and peregrine falcon are listed as endangered. The woodland

Table 2. Species on the Blue List in Alberta

grizzly bear
northern long-eared bat
Ord's kangaroo rat
red-tailed chipmunk
woodland caribou
wolverine
bay-breasted warbler
black-throated green warbler
Cape May warbler
ferruginous hawk
long-billed curlew
sage grouse
short-eared owl
Sprague's pipit
trumpeter swan
plains spadefoot toad
spotted frog
prairie rattlesnake
short-horned lizard
western hognose snake

caribou, barren-ground caribou, northern leopard frog, trumpeter swan, ferruginous hawk, burrowing owl, and piping plover are listed as threatened.

The Wildlife Act prohibits hunting, harassing, capturing, injuring, or killing and disturbing the nests or dens of endangered animals (but not endangered invertebrates or fish) which are listed in a schedule to the Act. If convicted of these acts, a person is liable to a fine of up to $100,000 or 6 months in jail.

The Wildlife Act also allows the Minister to make regulations protecting and restoring wildlife habitat, including that of endangered species. However, these provisions have never been used and there is no habitat in Alberta that is legally protected for the benefit of endangered species. In fact, the provincial government insists that Alberta's attempt at a protected areas strategy, Special Places 2000, is not about protecting wildlife, but only about representing landscapes.

In 1996, a few months before it signed the National Accord, the Alberta government amended the Wildlife Act to add the following provisions dealing with endangered species:

- the Minister *must* establish a committee known as the "Endangered Species Conservation Committee";
- the Committee is to *advise* the Minister about endangered species and make *recommendations* about:
  - the preparation and adoption of recovery plans;
  - species that should be established as endangered; and
  - endangered species and biodiversity conservation;
- the Committee *must* establish an independent scientific committee to *recommend* species that should be established as endangered;
- endangered species recovery plans *may* include population goals and the identification of critical habitats and strategies for population recovery. [italics added].

Thus, the only things that *must* happen are the establishment of the committee and the scientific subcommittee. The listing of species, the implementation of recovery plans, and the identification (and protection) of critical habitats, each of which are an obligation under the National Accord, are entirely at the discretion of the Minister of Environmental Protection.

The Endangered Species Conservation Committee includes representatives from the Alberta Cattle Commission, the Canadian Association of Petroleum Producers, the Alberta Forest Products Association, the Western Stockgrower's Association, the Alberta Fish and Game Association, the Alberta Native Plant Council, the Calgary Zoological Society, the Alberta Chapter of the Wildlife Society, First Nations, academic representatives, and the Alberta Conservation Association. No major conservation advocacy group sits on the committee. This is in part because the only group to be invited, the Federation of Alberta Naturalists, has refused to participate until the process is adequately funded. In the 2.5 years since these amendments were passed, the committee has met only twice and has not yet recommended any species to be designated as endangered. The scientific committee is still being established.

WHERE'S THE BEEF?

Alberta government officials maintain the province is taking a "cooperative approach" that will engage landowners and those managing Crown lands in the recovery process voluntarily (Alberta Environmental Protection staff, 1999, pers. comm.). However, the process is a painfully slow way of dealing with what is generally understood to be a deepening crisis.

Further, voluntary efforts to achieve environmental protection goals will be effective only if legislation is in place to authorize the voluntary efforts, monitor their effectiveness, and enable a regulatory response if voluntary initiatives are failing. The Alberta approach does none of these things.

The insistence of the Alberta government and others that the voluntary cooperation of landowners and managers is the only way to implement protection for endangered species is puzzling. Why do the interests of ranchers and others responsible for managing land take priority over those of species teetering on the brink of extinction? Some of those who say that they want only voluntary programs to protect endangered species habitats also say they will kill endangered species and destroy their habitat if habitat protection laws are passed. What kind of noble caretakers are these?

These kinds of threats would not be tolerated in other societal spheres. Imagine if we relied on "voluntary cooperation" for the payment of taxes, for compliance with zoning bylaws, or for maintaining safe highway speeds. These are all

regulatory schemes that affect how we deal with private property; as law-abiding citizens we accept them. Should we expect no less for regulations to deal with the crisis of species extinctions?

As a society, we make lofty promises to protect the environment. But we do a dismal job of bringing that commitment to the ground and making some of the difficult trade-offs needed if we are to have a truly sustainable future. Governments and industries support protected areas, but only if they don't prevent resource extraction. We seem to think that we can have our cake and eat it too. We want to protect species and habitats without having to give up any of our consumptive activities. Protecting endangered species is one of the highest issues on the public and government agendas, yet we choose not to stop the activities that are causing the extinction crisis.

CONCLUSION

Albertans want endangered species to be protected . The government has said it will protect them and meet the requirements of the National Accord. Yet, this is not being done. The effectiveness of Alberta's approach depends on the political will of the Minister of Environmental Protection and his Cabinet colleagues. This approach does not ensure that every effort is made to secure the recovery of endangered species. Rather, it guarantees trade-offs will continue to be made that favour economic development and private interests over species protection.

Endangered species legislation is the mechanism of last resort when our other efforts—sustainable management of the landscape, protected core habitats, and voluntary initiatives—have failed. Given the severity of the crisis, and given that endangered species legislation applies only when all other efforts have failed, this legislation must be strong and clear so that the causes of a population's decline can be addressed quickly and effectively. Sometimes, such actions will mean some people or companies will be restricted on their own or Crown land. In such rare cases, a process for compensation is appropriate.

Our attitude toward the other creatures with which we share this beautiful planet is the ultimate test of our success as a species. Are we so arrogant that we are unwilling to temper our own behaviour so that other species can survive? Or are we generous and intelligent enough to curb our activities so the survival of all species, including our own, can be assured?

Alberta's endangered species provisions do not demonstrate that commitment. They leave the critical decisions of listing, habitat protection, and recovery plan implementation to the realm of political will where economic considerations can prevail. I think we can, and must, do better than that.

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Conserving Mammals at Risk: The Role of Taxonomy

David W. Nagorsen
Royal British Columbia Museum
P.O. Box 9815, STN PROV GOVT, Victoria, BC, V8W 9W2, Canada
Dnagorsen@RBML01.rbcm.gov.bc.ca

Mark A. Fraker
TerraMar Environmental Research Ltd.
8617 Lochside Drive, Sidney, BC, V8L 1M8, Canada

Nick Panter
Royal British Columbia Museum
P.O. Box 9815, STN PROV GOVT, Victoria, BC, V8W 9W2, Canada

ABSTRACT

A number of mammalian species, subspecies, and populations are designated at risk both nationally and by the province of British Columbia. Despite the perception that the taxonomy of mammals is well known, major taxonomic problems are associated with some of these taxa. Species limits and reliable identification traits need to be determined for several problematic species groups. Moreover, the validity of many of the subspecies listed at risk is unknown. Some were described more than 50 years ago from small samples and descriptive methodology, and modern research is essential to determine if they represent distinct evolutionary units that warrant protection. We use recent research on listed subspecies of chipmunks (*Tamias* spp.), northern pocket gophers (*Thomomys talpoides*), and southern red-backed voles (*Clethrionomys gapperi*) from the Kootenay region of British Columbia to illustrate how taxonomy is critical for delimiting conservation units. Preliminary results suggest that the 2 subspecies of the red-tailed chipmunk, *Tamias ruficaudus* (*T. r. simulans*, *T. r. ruficaudus*) are full species. Similarly, the 2 allopatric subspecies of the least chipmunk, *Tamias minimus* (*T. m. selkirki*, *T. m. oreocetes*) appear to be strongly differentiated from other populations, warranting recognition as distinct evolutionary units. In contrast, the Wynndel subspecies (*T. t. segregatus*) of the northern pocket gopher is probably a localized population of *T. t. saturatus* with minor differentiation attributable to genetic drift. Gale's race (*C. g. galei*) of the southern red-backed vole is even more dubious and demonstrates the problems in delimiting discrete units in widespread species with continuous distributions. Comparable modern studies could be applied to most of the subspecies currently listed at risk. A high priority provincially should be the allopatric endemic taxa that inhabit coastal islands or isolated mainland habitats.

Key words: *Clethrionomys gapperi*, least chipmunk, northern pocket gopher, red-tailed chipmunk, southern red-backed vole, *Tamias minimus*, *Tamias ruficaudus*, taxonomy, *Thomomys talpoides*.

Taxonomy, the theory and practice of classifying organisms (Mayr and Ashlock 1991), plays a central role in conserving biodiversity and species at risk. The most important contribution of taxonomy to conservation biology is defining the taxonomic units that we are trying to protect, such as species or subspecies. The definition and biological concepts of species and subspecies and their value as evolutionary significant units (ESUs) in conservation biology has promoted considerable debate among taxonomists and conservation biologists (O'Brien and Mayr 1991, Rojas 1992, Engstrom et al. 1994).

Nationally, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has designated 30 mammalian taxa (18 species, 12 subspecies or populations) as either Endangered, Threatened, or Vulnerable (COSEWIC 1999). According to the most recent Red and Blue lists released by the British Columbia Ministry of Environment, Lands and Parks (Cannings et al. 1999), the province recognizes 49 mammalian taxa at risk in British Columbia: 22 species and 27 subspecies. Although mammals as a group are generally regarded as well understood taxonomically, much of the taxonomy supporting these listings is dated and largely the result of research done 50 to 100 years ago. The number of valid species is unclear for several groups such as the long-eared myotis bats (*Myotis evotis*, *M. keenii*, *M. septentrionalis*), and the chipmunks (*Tamias* spp.). Taxonomy at the subspecies level is even more uncertain. COSEWIC's national designations of even high-profile, large mammals such as the gray wolf (*Canis lupus*), grizzly bear (*Ursus arctos*), and caribou (*Rangifer tarandus*) have been hampered by taxonomic controversies. The 27 mammalian subspecies listed by the province are also problematic, with most described from subjective methods using

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Table 1. Red and Blue List status of 6 rodent subspecies at risk from the Kootenay region of southeastern British Columbia.

Common name	Scientific name	Status
Red-tailed chipmunk (Rocky Mountain ssp.)	*Tamias ruficaudus ruficaudus*	Red
Red-tailed chipmunk (Selkirk ssp.)	*Tamias ruficaudus simulans*	Red
Least chipmunk (Timberline ssp.)	*Tamias minimus oreocetes*	Red
Least chipmunk (Selkirk ssp.)	*Tamias minimus selkirki*	Red
Northern pocket gopher (Wynndel ssp.)	*Thomomys talpoides segregatus*	Red
Southern red-backed vole (Gale's ssp.)	*Clethrionomys gapperi galei*	Blue

inadequate sample sizes. Their validity as distinct evolutionary significant units that should be protected and conserved is unknown. A few of the listed subspecies may be full species but most are dubious.

Many of these taxonomic problems are the legacy of early taxonomic research on North American mammals. Early taxonomists operated with a typological species concept. They had little appreciation of geographic variation, and virtually every mammalian population with minor differences was described as a new species (Engstrom et al. 1994). This was followed by taxonomists adopting the polytypic species concept (Mayr and Ashlock 1991), where species are recognized as being geographically variable, and distinct geographic forms are formally treated under the Code of Zoological Nomenclature as subspecies with a trinomen. This led to numerous taxonomic revisions, with many species being reduced to the subspecies level. However, in most of these revisions taxonomists tended to retain the original species names as subspecies. Often "species" were simply reduced to subspecies with no rigorous studies of geographic variation to substantiate that they were in fact taxonomically different. Many of these subspecies persist today and are listed in standard synoptic references such as Cowan and Guiguet (1965), Banfield (1974), Hall (1981), and Nagorsen (1990). Whatever conceptual basis is applied to define species or subspecies, modern taxonomic classifications are based on detailed studies of geographic variation applying rigorous quantitative techniques with morphological and molecular data. Few such modern studies have been done on Canada's mammalian fauna.

In this paper we review results from our taxonomic research on 6 subspecies representing 4 provincially listed rodent species (Table 1) from the Kootenay region of southeastern British Columbia. We use these taxa as case studies to explore some of the issues concerning taxonomy and conservation. It should be emphasized that our results are based on small samples and are preliminary.

METHODS

In 1996, the Columbia Basin Fish and Wildlife Compensation Program, the Ministry of Environment, Lands and Parks, the Royal British Columbia Museum, and Terramar Environmental Research Ltd. initiated a collaborative study to investigate the taxonomy, distribution, and conservation status of 6 taxa of small mammals from the Kootenay region. Taxonomic analyses are based on voucher specimens collected from 1996 to 1998. In 1996, voucher samples included southern red-backed voles (*Clethrionomys gapperi*) and chipmunks (*Tamias* spp.) from the Akamina-Kishinena region of the southern Rocky Mountains, chipmunks from the southern Selkirk Mountains, and northern pocket gophers (*Thomomys talpoides*) from the Creston Valley. The 1997 field studies concentrated on chipmunks from the Purcell Mountains and adjacent areas in the Rocky Mountain trench; 1998 field studies concentrated on chipmunk samples from the southern Rocky Mountains and Purcell Mountains. Detailed information on sampling sites and vouchers specimens were summarized by Nagorsen (1997), Fraker and Nagorsen (1998), and Nagorsen and Fraker (1999). Our voucher samples were augmented with historical museum specimens taken in the Kootenay region.

Our analyses were heavily based on genital bone morphology, an important taxonomic and identification trait in chipmunks. All voucher specimens were identified to species from genital bone preparations. Identification of historical museum specimens was verified from radiographs of study skins that revealed genital bones. Canonical variate analyses based on 8 male genital bone measurements and 15 cranial measurements were used to assess geographic variation. Samples of the red-tailed chipmunk (*T. ruficaudus*) were too small for rigorous statistical analysis, and only results for the least chipmunk (*T. minimus*) are shown. For details on the taxonomic analyses of chipmunks see Fraker and Nagorsen (1998) and Nagorsen and Fraker (1999).

Nagorsen (1997) summarized the techniques used to assess variation in skull shape and pelage colour of *T. talpoides*. We analyzed 10 samples representing 7 subspecies from western Canada. Results are shown for females. Because size is highly plastic in pocket gophers we removed size effects from the data with the Burnaby shearing technique (Rohlf and Bookstein 1987) and examined patterns of variation in skull shape with a canonical variate analysis of 8 skull measurements. Studies on a related species in California have shown that patterns of variation defined by skull shape may be concordant with genetic variation (Smith

and Patton 1988). We scored the degree of white spotting of the pelage from 4 ventral regions of museum study skins: chin, chest, forearms, and abdomen. Each area was ranked from 0 (no spotting) to 3 (heavy spotting) for the degree of spotting and scores were totalled for an overall spotting index of each specimen.

Our analysis of *C. gapperi* was restricted to a superficial comparison of our vouchers with historical museum specimens and a review of the published descriptions for the 8 subspecies recognized in British Columbia. No quantitative analyses were done.

CASE STUDIES

Red-tailed Chipmunk (*Tamias ruficaudus*)

T. ruficaudus is an example of a species-level problem. Two subspecies are currently recognized: *T. r. ruficaudus* associated with the Rocky Mountains of Montana, Idaho, and southern British Columbia; and *T. r. simulans* associated with the Selkirk Mountains of eastern Washington, northern Idaho, and southern British Columbia (Fig. 1). From a study of baculer (i.e., male genital bone) morphology in populations from Washington, Idaho, and Montana, Patterson and Heaney (1987) suggested that the 2 taxa are full species. Preliminary results of an analysis of our voucher specimens and known museum specimens from western Canada demonstrate that at the northern periphery of their range in Canada, the 2 subspecies differ in pelage, skull, and genital bone morphology. The 2 taxa also show ecological differences, with *T. r. ruficaudus* confined to the subalpine (1,800–1,900 m) in the Rocky Mountains, whereas the known range of *T. r. simulans* is from 700 to 1,830 m in the Selkirk Mountains. Cowan and Guiguet (1965) and Hall (1981) showed the 2 subspecies in contact in the Purcell Mountains and Rocky Mountain trench. However, recent field studies and an examination of historical museum specimens have revealed that *T. ruficaudus* is absent from the Purcell Mountains. Thus, the distributions of *T. r. ruficaudus* and *T. r. simulans* in Canada appear to be allopatric, with the 2 taxa separated by the Creston Valley, the Rocky Mountain trench, and the intervening Purcell Mountains (Fig. 1). All of these data support the recognition of *T. r. ruficaudus* and *T. r. simulans* as full species. Molecular research now being done by Jack Sullivan in Idaho where the 2 groups may be in contact should resolve the species question.

Both *T. r. ruficaudus* and *T. r. simulans* are listed on the provincial Red List at the subspecies level, and, from a conservation perspective, debates about differentiation at the species versus subspecies level may appear academic. However, a listed species has higher conservation priority than a listed subspecies. More importantly, most of what is known of the biology of *T. ruficaudus* is based on studies of *T. r ruficaudus* done in Montana (Beg 1969) and Beg's data may not be relevant to *T. r. simulans*, particularly if it is a different species.

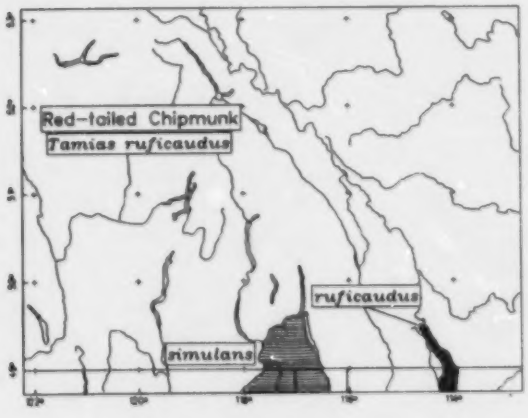

Figure 1. Distribution of the 2 subspecies of the red-tailed chipmunk (*Tamias ruficaudus*) in Canada.

Least Chipmunk (*Tamias minimus*)

T. minimus is represented by 4 subspecies in British Columbia (Fig. 2). The northern subspecies, *T. m. borealis* and *T. m. caniceps*, are widespread; in contrast, *T. m selkirki* and *T. m. oreocetes* are alpine races found above 1,900 m. *T. m. selkirki* was described and named by Cowan (1946) from only 5 specimens taken from the type locality at Paradise Mine in the Purcell Mountains. Our field surveys in 1997 (Fraker and Nagorsen 1998) and 1998 (Nagorsen and

Figure 2. Distribution of the 4 subspecies of the least chipmunk (*Tamias minimus*) in British Columbia and western Alberta.

Fraker 1999) added a few new locations, but this taxon is still known from only a very restricted area. *T. m. oreocetes*, a small pale form described by Merriam (1897), occupies the Rocky Mountains of southern Canada and northern Montana. Taxonomic validity of these races is not clear and the precise distribution of *T. m. oreocetes* in Canada is contentious. Cowan (1946) concluded that it was restricted to the Waterton Lakes area in Canada, but Crowe (1943) assigned *T. minimus* from as far north as Mount Assiniboine Provincial Park and Tornado Pass to *T. m. oreocetes*, a hypothesis supported by Banfield (1958).

Our analyses demonstrate that *T. m. selkirki* is strongly differentiated from both *T. m. oreocetes* and *T. m. borealis* in male bacular bone morphology (Fig. 3). A similar pattern of variation was found in our analysis of cranial morphology. The strong differentiation of *T. m. selkirki* from *T. m. oreocetes*, despite their geographic proximity, is significant. These 2 alpine populations, separated by the Rocky Mountain trench and lowland populations of the yellow pine chipmunk (*T. amoenus*), have probably been isolated since the early postglacial. Although *T. m. oreocetes* also appears to be morphologically distinct, its taxonomic status and distribution are unresolved and more samples are needed. The small, pale form of *T. minimus* in the southern Canadian Rocky Mountains may represent 1 extreme in a north-south cline that extends along the Canadian Rocky Mountains. Cowan (1946) described clinal variation in body measurements among northern *T. m. borealis* and the southern *T. m. oreocetes*. However, if *T. m. borealis* and *T. m. oreocetes* are separated by the Bow River and lowland populations of *T. amoenus* in Banff National Park, as suggested by Banfield (1958), then it is possible that a sharp step-cline delimiting these 2 subspecies could occur across the Bow River. Our results are based on a sample of only 3 *T. m. oreocetes* specimens from Alberta. A comprehensive study based on all historical museum specimens and new material taken in August 1998 is in progress.

The extent to which these chipmunk subspecies are at risk is not clear. If taxonomic studies demonstrate that *T. m. oreocetes* ranges as far north as the Bow River in Canada, then much of the Canadian range of this taxon falls in national and provincial parks and it may be appropriate to downlist its ranking. Because its known distributional area is small and isolated from other populations, *T. m. selkirki* is more likely to be at risk and it was recently listed as Vulnerable by the International Union for the Conservation of Nature (IUCN; Sullivan and Nagorsen 1998).

Northern Pocket Gopher (*Thomomys talpoides*)

A fossorial mammal that spends much of its life underground, the distribution of *T. talpoides* is fragmented by areas of unsuitable soil and physical barriers such as rivers and lakes. The pocket gophers are the most morphologically and genetically variable of all mammals and some 58 subspecies have been described for *T. talpoides*, most defined by minor differences in fur colour (Hall 1981). This excessive naming of every local variant in pocket gophers has been singled out as an example of a misuse of the subspecies concept (Simpson 1961).

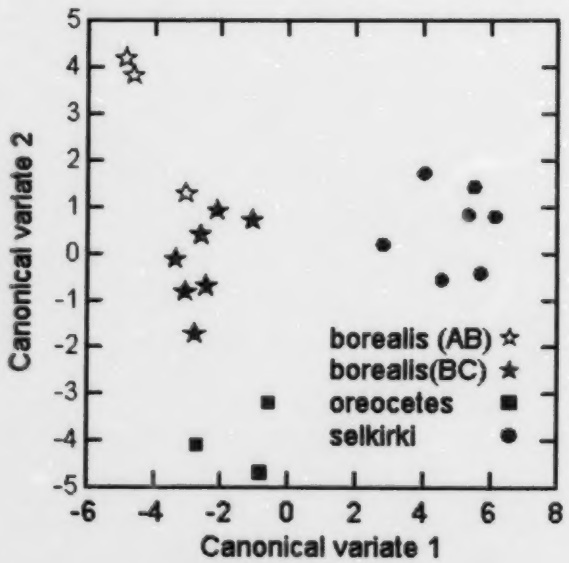

Figure 3. Projection of 4 groups of least chipmunks (*Tamias minimus*) representing 3 subspecies on the first 2 canonical variates derived from 8 bacular bone measurements.

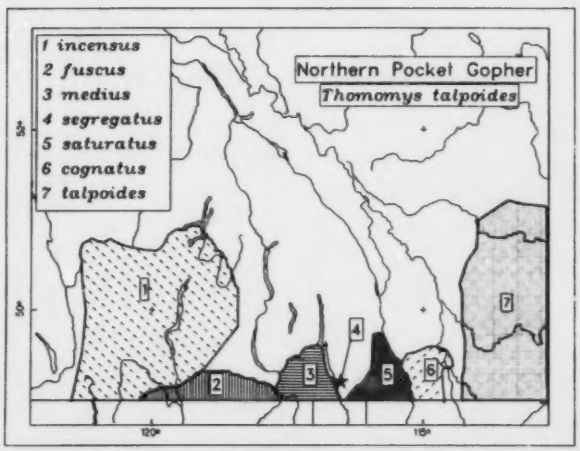

Figure 4. Distribution of 7 subspecies of the northern pocket gopher (*Thomomys talpoides*) in British Columbia and western Alberta.

Taxonomy of the British Columbian populations was last revised by Johnstone (1954). Using largely descriptive methodology, he recognized 6 subspecies (Fig. 4) including 2 previously undescribed subspecies, and argued that they were maintained by the isolating effects of the Columbia and Kootenay rivers. Based on a comprehensive inventory, Fraker et al. (1997) found that *T. t. segregatus*, the isolated race found near Wynndel above the Kootenay floodplain north of Creston, still occupies the same 10-km^2 area that it did 50 years ago when first discovered by Munro (1950). Even though *T. t . saturatus* invaded the southern Creston Valley from Idaho in the 1960s, *T. t . saturatus* and *T. t . segregatus* are allopatric in British Columbia, with their distributions separated by about 13 km (Fraker et al. 1997). Because of its small population size and isolation, *T. t. segregatus* is susceptible to extinction from stochastic events. However, the fundamental issue is whether this population represents a distinct taxonomic unit or is simply a local variant of an adjacent subspecies such as *T. t. saturatus* or *T. t. medius*, with minor differences that can be attributed to genetic drift in a localized population.

Our analyses of 10 samples representing 7 subspecies from western Canada, demonstrated that the prairie race *T. t. talpoides* from southwestern Alberta falls out as a distinct group (Fig. 5). The 6 races from British Columbia overlap on the first axis, but there is some separation on the second axis, with samples from the Kootenays at the top and samples from the Okanagan-Cascades at the bottom. Nonetheless, the morphological data provide little evidence for 6 distinct groups in the province and, most significantly, the Wynndel subspecies *T. t. segregatus* does not fall out clearly as a distinct form.

The incidence of the white spotting mutation is variable within and among the 10 populations (Fig. 6) with no strong concordance with subspecies. However, this coat colour mutant is most prevalent in *T. t. segregatus*. The strong differences in this mutation exhibited by the Wynndel population and the sample of *T. t. medius* from the west side of the Creston Valley is striking and supports the hypothesis of Johnstone (1954) and Fraker et al. (1997) that the Kootenay River and floodplain form a formidable barrier to pocket gopher dispersal and gene flow.

Although the use of subspecies is controversial, particularly for highly variable taxa such as pocket gophers, Smith and Patton (1988) supported the concept if the subspecies reflect evolutionary units that share similar morphological and genetic traits. Despite their support for subspecies, Smith and Patton (1988) recommended that local variants attributable to genetic drift in small populations do not warrant subspecific status. *T. t. segregatus* could be a relict population resulting from an early postglacial invasion of British Columbia or it may be a recent isolate of *T. t . saturatus*,

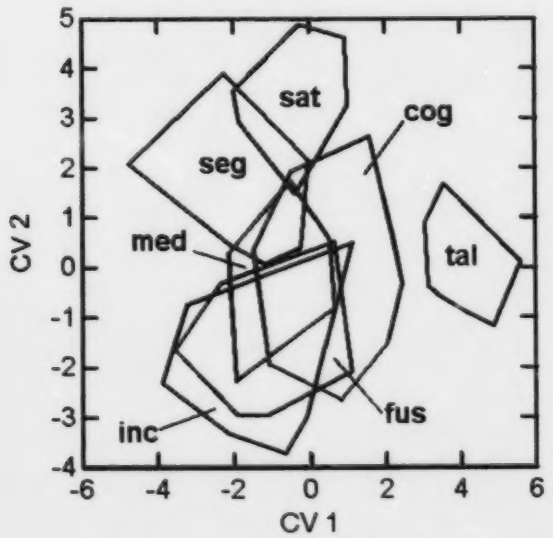

Figure 5. Projection of 7 subspecies of the northern pocket gopher (*Thomomys talpoides*) on the first 2 canonical variates derived from 8 size-corrected skull measurements. Polygons enclose the outermost points of each subspecies. (cog = *cognatus*; fus = *fuscus*; inc = *incensus*; med = *medius*; sat = *saturatus*; seg = *segregatus*; tal = *talpoides*.)

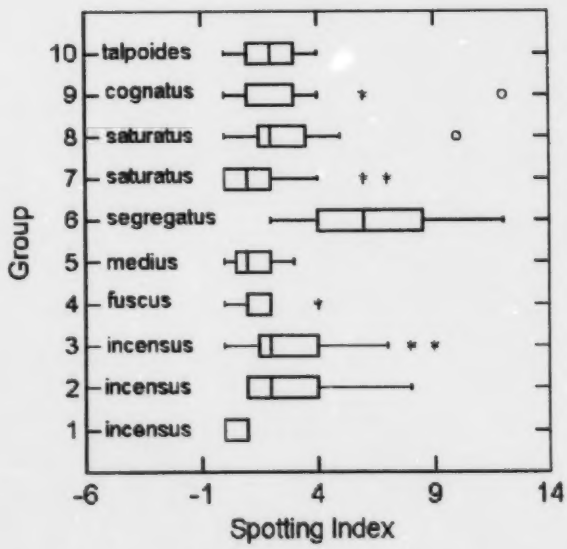

Figure 6. Box plots showing the degree of white spotting in 10 samples representing 7 subspecies of the northern pocket gopher (*Thomomys talpoides*). Vertical line is the median; the box delimits the hinges (1st and 3rd quantiles); stars and open circles indicate outliers.

with most of its divergence (the white spotting and minor differentiation in skull morphology) attributable to drift in a small isolated population. Only genetic studies (chromosomes, allozymes, DNA) will clarify the taxonomy of *T. talpoides*. If it is revealed that *T. t segregatus* is not a valid subspecies and it is synonymized with another race, such as *T. t. saturatus*, which is not at risk, then *T. talpoides* would simply disappear from the provincial Red List. However, until such taxonomic studies are done it is prudent to maintain *T. t segregatus* on the provincial Red List. In a recent IUCN conservation action plan for North American rodents this subspecies was designated as "Lower Risk, near threatened" with a recommendation for a modern taxonomic study of the species (Yensen and Nagorsen 1998).

Southern Red-backed Vole (*Clethrionomys gapperi*)

C. gapperi illustrates the problem of delimiting subspecies in species with a continuous distribution. Eight subspecies are recognized in the province (Fig. 7). *C. g. galei*, a widespread race found in the Rocky Mountains of the United States, barely enters the southeast corner of the province. It is appears on the provincial Blue List, presumably because of few locality records, although the species seems abundant and widespread in the southern Canadian Rocky Mountains (Fraker and Nagorsen 1998). Another subspecies, *C. g. occidentalis*, occupies the Puget Sound lowlands of Washington Sate and the extreme southwestern edge of the Fraser Valley. Known in the province from only 2 historical records in the Vancouver area, this taxon is Red-listed and may be extirpated in British Columbia. However, there has been no modern taxonomic revision of *C. gapperi* and most of the named subspecies are a legacy of early species descriptions. Both

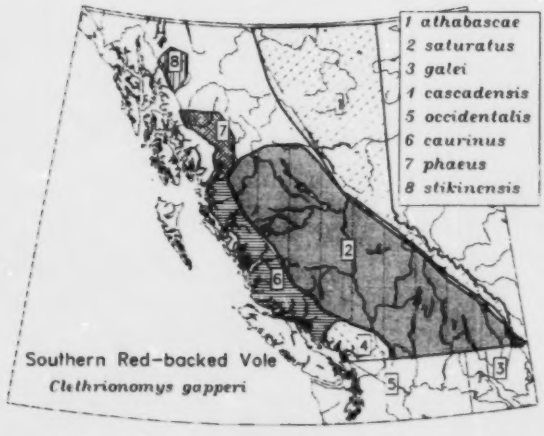

Figure 7. Distribution of the 8 subspecies of the southern red-backed vole (*Clethrionomys gapperi*) in British Columbia.

C. g. galei and *C. g. occidentalis* were originally described as species by Merriam (1890) under the old generic name *Evotomys* based on vague descriptive traits taken from only a few specimens. For example, Merriam's description of *Evotomys* (= *Clethrionomys*) *galei* as a new species was based solely on a single specimen—the type specimen taken from Boulder Colorado. His original description consists of vague statements such as: "size about equal to that of *E. gapperi*, or larger,....above considerably lighter than true *gapperi*,...the molar series are considerably larger than in skulls of *gapperi* of the same size..." The last major taxonomic revision of North American forms of *Clethrionomys* was done more than 100 years ago by Bailey (1897). The group is clearly in need of a modern taxonomic revision.

As part of the Kootenay small mammal project we were asked to determine the validity of *C. g. galei* using the few voucher specimens taken during the 1996 inventory and existing museum specimens from the Kootenay region. However, to resolve the subspecies taxonomy of *C. gapperi* requires a major taxonomic study across western North America, assessing geographic variation in morphological and genetic traits. Several distinct taxonomic units of *C. gapperi* probably occur in British Columbia, but it is unlikely that they will be concordant with the 8 subspecies currently recognized. Because it is abundant, ecologically widespread, and contiguous with other subspecies such as *C. g. saturatus*, we recommend that *C. g. galei* be removed from the provincial list of vertebrates at risk. *C. g. occidentalis* and other coastal subspecies (*phaeus*, *stikinensis*, *caurinus*, *cascadensis*) probably represent a single coastal evolutionary unit. However, until the taxonomy is revised, it is appropriate to continue to treat *C. g. occidentalis* as a taxon at risk.

DISCUSSION

The small mammals from the Kootenay region selected for our study support our hypothesis that many of the taxonomic units of mammals currently listed at risk are the product of dated and invalid taxonomy. The *T. ruficaudus* example demonstrates that species-level problems still exist with some Canadian mammals. Our support for recognizing the 2 subspecies (*T. r. ruficaudus* and *T. r. simulans*) as full species is consistent with the prediction of Engstrom et al. (1994) that the number of North American mammal species will increase with additional taxonomic research. Evidently a number of sibling species were reduced to the subspecies level in the wave of taxonomic revisions that followed the use of the polytypic species concept. Most taxonomic problems for listed mammals are associated with subspecies and populations. Modern taxonomic techniques will undoubtedly confirm the validity of some of these taxa. Despite the small sample sizes and descriptive methodology used by Cowan (1946), our

findings support Cowan's conclusions and strengthen the case for treating *T. m. selkirki* as a distinct taxonomic unit of the least chipmunk. However, many subspecies, particularly in species with continuous distributions such as *C. gapperi*, are of dubious taxonomic validity. The trend over the past few decades has been a reduction in the number of accepted subspecies for North American mammals (Engstrom et al. 1994). Modern taxonomic revisions may reduce the lists of subspecies at risk through synonomy.

Although most of our focus has been on verifying taxa already named and described, it is conceivable that there are well-defined taxonomic units of mammals that have yet to be identified. Limited data on yellow-pine chipmunks (*Tamias amoenus*) from the Kootenay region suggest that populations on the west and east sides of the Rocky Mountain trench differ strikingly in male bacular bone morphology (Fraker and Nagorsen 1998). Although populations from the Kootenay region are currently assigned to a single subspecies, *T. a. luteiventris* (Cowan and Guiguet 1965), this taxon may consist of several strongly differentiated groups. Thus significant taxonomic diversity may be concealed within some recognized taxa.

Resolving these taxonomic problems is a major task that must be done on a species by species basis by applying powerful modern tools including molecular techniques and numerical analysis of traditional morphological traits. However, with few funds and taxonomists available, little taxonomic research is being done on Canada's mammalian fauna. In 1993 Canada's national museum, the Canadian Museum of Nature, eliminated its mammalogist position and research program in mammalian systematics. Provincially in British Columbia, the Royal British Columbia Museum struggles with tight budgets and limited resources. No mammalian taxonomists are on staff at any of the province's universities. Provincial funding for the province's Red- and Blue-listed mammals has been directed at field inventory and habitat studies rather than verification of the taxonomic validity of the listed taxa.

A taxonomic assessment of every mammalian taxa listed nationally or provincially is clearly not feasible or warranted. Provincially, a high priority should be the allopatric, endemic taxa. Endemic taxa warrant special concern because they occur nowhere else. Allopatric populations are more likely to be differentiated taxonomically because of isolation and restricted gene flow. This isolation also makes them vulnerable to extinction because there is little opportunity for a rescue effect. A few allopatric, endemic mammalian taxa (e.g., *T. m. selkirki*) occur in isolated mainland habitats of British Columbia, but most are associated with the numerous coastal islands (Nagorsen 1990). Research now being done on the insular mammals of southeast Alaska (MacDonald and Cook 1996) is a model for the kind of research that needs to be done in British Columbia.

ACKNOWLEDGEMENTS

Our research was supported by the Royal British Columbia Museum, the Columbia Basin Fish and Wildlife Compensation Program, Forest Renewal British Columbia, and the British Columbia Ministry of Environment, Lands and Parks. We thank J. Krebs, T. Antifeau, and L. Ingham for their strong support. Our chipmunk research benefited from discussions with I. McTaggart Cowan. The Canadian Museum of Nature; Cowan Vertebrate Museum, University of British Columbia; Museum of Zoology, University of Alberta; Provincial Museum of Alberta; and Royal Ontario Museum provided loans of chipmunk specimens and allowed one of us (DWN) to measure pocket gopher specimens in their collections.

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Species at the Edge: The Case for Listing of "Peripheral" Species

David F. Fraser

Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9374, STN PROV GOVT, Victoria, BC, V8W 9M4, Canada
Dave.Fraser@gems8.gov.bc.ca

ABSTRACT

Peripheral species are those that are at the edge of their range, often barely crossing the line into a political jurisdiction. Formerly, the process for listing species in British Columbia prevented peripheral species from becoming candidates for threatened and endangered status. Reasons for changing this policy include patterns of species collapse, historical importance of peripheral species in recovery planning, and genetic, evolutionary, and biodiversity considerations.

Key words: peripheral species, species collapse.

Peripheral taxa are those that are at the edge of their range, usually those that barely extend into a political jurisdiction. Since the late 1980s, British Columbia has risk-listed species of vertebrates, with Red-listed species being of the greatest conservation concern, followed by Blue-listed, and lastly by Yellow-listed species (Harcombe 2000). Before 1992, the British Columbia Wildlife Branch placed species at risk that were considered peripheral on the Blue List. Those peripheral species that were candidates for threatened or endangered status were down-listed from Red to Blue. This policy was designed to ensure that scarce resources were not spent on species that could not be recovered because they were limited by climate or habitat factors beyond human control.

This policy was changed for several reasons:

1. Many endangered species collapse to the periphery of their range. Lomolino and Channell (1995), examined 31 endangered mammal species, 23 of which had extant populations along the historic periphery of their range and had been extirpated in the core of their range. Lomolino and Perault (1998) examined 245 species that had collapsed to <25% of their historic range, most collapsed to the periphery of their range. They concluded that "sites along the periphery of a species' historic range should no longer be dismissed as sites with little conservation value." This trend is particularly important for British Columbia, as the most common pattern (for mammals) in the northern hemisphere is to collapse to the northern and western edges of a species' range (Lomolino and Channell 1995).

   The pattern of collapsing to the edge of a species' range is not restricted to vertebrates. For example, the golden paintbrush (*Castilleja levisecta*) was formerly distributed from southern Vancouver Island south through the Willamette Valley to central Oregon (Peck 1961, Douglas and Ryan 1999). Currently, this species is restricted to 7 sites in Puget Sound, Washington (Sheenan and Sprague 1984) and some small islands off southern Vancouver Island, near the northern edge of the species' historic range (Douglas and Ryan 1999).

   History has shown that peripheral populations can be important in conservation efforts. In some cases peripheral populations are the only populations that are healthy enough to provide animals for reintroduction or recolonization. Sea otters (*Enhydra lutra*) reintroduced along the Pacific Coast were taken from peripheral populations that escaped overharvesting (Watson et al. 1997). California bighorn sheep (*Ovis canadensis californiana*) taken from healthy populations at the northern edge of their range in British Columbia have been used in reintroductions in areas of the western United States (Charles and Leslie 1999). Columbia sharp-tailed grouse (*Tympanuchus phasianellus columbianus*) found at the very northern edge of this subspecies' range provide the only healthy populations of a bird that once extended from the Cariboo to California, and are being used to repopulate sites in the United States (M. Chutter, Wildlife Branch, B.C. Ministry of Environment, Lands and Parks, pers. comm.). Within hours of the confirmation of the identification of a population of Oregon spotted frogs (*Rana pretiosa*) in British Columbia, there were requests from the United States for egg masses for genetic analysis (L. Friis, Wildlife Branch, B.C. Ministry of Environment, Lands and Parks, pers. comm.).
2. Populations at the geographic margins of their ranges may be important for the long-term survival and evolution of species, as they are major contributors to evolutionary

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

change (Mayr 1982, Lesica and Allendorf 1995).

Founder effects and drift due to reduced gene flow should cause peripheral populations to have different gene frequencies than central populations. This has been well documented for plants (Lesica and Allendorf 1995). Scudder (1989) stated, "Marginal populations have a high adaptive significance to the species as a whole, and marginal habitat conservation, preservation and management is one of the 'best' ways to conserve the genetic diversity...". Many consider the loss of genetically distinct populations to be as important as the loss of species (Ehrlich 1988, Ledig 1993).

Here lies an interesting conservation conundrum that has been explored by Noss (1994). Disjunct or peripheral populations of species can be expected to be more genetically impoverished than central populations—but also genetically distinct from central populations, because of reduced gene flow to these isolated or marginal populations. The pattern presents a conservation dilemma because populations with lower heterozygosity are likely to be less adaptable to future environmental change (Frankel and Soule 1981) and therefore might be seen as less important to conserve. Peripheral populations are also likely to be in suboptimal habitat, making conservation more effective when directed to the central portion of each species' range. However, disjunct or peripheral populations are likely to have diverged genetically from central populations due to either genetic drift or adaptation to local environments. Selective pressures can be expected to be intense for these populations. If we are concerned with maintaining opportunities for speciation and future biodiversity, then the conservation of peripheral and disjunct populations is critical (Noss 1994).

Presumably because of these populations adapting to selective pressures, species at the edge of their range can occur in unusual or atypical habitats; for example, flammulated owls (*Otus flammeolus*) in British Columbia use different forest types than in other parts of their range (van Woudenberg and Kirk 1999); and 1 of the northernmost Garry oak (*Quercus garryana*) stands in British Columbia has an understory comprised of wild ginger (*Asarum caudatum*)—a species usually associated with wet, coniferous forests (G. Sirk, Comox-Strathcona Regional District, pers. comm.). Certain uncommon genotypes may be well represented in peripheral populations. For example, the isolated populations of slender-footed waterleaf (*Hydrophyllum tenuipes*) in British Columbia are predominately purple-flowered (Fraser in press)—a relatively rare flower colour for this species in the core of its range in the United States (Pojar and MacKinnon 1994).

3. Some species at risk at the edge of their range are also at risk over a large portion of their range. In British Columbia, for example, the Oregon spotted frog is at risk or extirpated throughout its historic range (literature reviewed in Haycock and Knopp in prog.); the coastal subspecies of gopher snake (*Pituophus melanoleuca catinifer*) is extirpated in Washington State (Storm and Leonard 1995); and the sage grouse (*Centrocercus urophasianus*) and sage thrasher (*Oreoscoptes montanus*) are both considered at risk in Washington State (Smith et al. 1997). The status of a species elsewhere should be considered before dismissing a peripheral population as unimportant.
4. Several high-level policy strategies call for the maintenance of all species in British Columbia (B.C. Ministry of Environment, Lands and Parks 1994, 1996) and in the Canadian Biodiversity Strategy (Biodiversity Convention Office 1995). Most recently, the National Accord for the Protection of Species at Risk committed signatories (the governments of Canada and most of the Canadian provinces and territories) to a program to protect species including "all native, wild species." These programs do not exclude species at the edge of their range.
5. The exclusion of peripheral taxa from legal protection and programs could result in a significant loss of Canada's genetic resources. Vertebrate species often take the role of umbrella species, because efforts made to protect the habitat of these species will likely help many other species. Efforts to protect species associated with 1 ecosystem type will protect many other species (some as yet unstudied) associated with that habitat type (Hunter and Hutchinson 1994). This applies to "peripheral" habitat types as well as (or perhaps better than) our "core" habitat types. In British Columbia, the small "at the edge of the range" habitats associated with the Peace River district, the Okanagan Valley, the Gulf Islands, the Lower Mainland, the southern Rocky Mountain Trench, and the Haines Triangle add much to the diversity of British Columbia. Since several of these "hot spots" for peripheral species are exactly those that have been selected for intensive human development because of their unusual climates, many of these species face a high level of threat from habitat loss.
6. Reintroductions of populations at the edge of a species' range are less likely to succeed than recovery efforts for populations at the core of a range (Griffith et al. 1989). Precautions for the conservation of peripheral populations may be more important given the poor prospect of recovery if eliminated or severely damaged.
7. Often species are more widespread than current knowledge would indicate. Spotted owls (*Strix occidentalis*), flammulated owls, spotted bats (*Euderma maculatum*), and tailed frogs (*Ascaphus truei*) in British Columbia have been found north of what was once considered their range when they were first listed. The inventories

required to document the extent of these species' ranges were not initiated until after they were "listed" and became of management concern. For tailed frogs, flammulated owls, and spotted bats this has resulted in both down-listing and changes in management practices (Cannings et al. 1999, Fraser et al. 1999). In other words, species thought to be at risk and "peripheral" may not be peripheral at all.

8. There is a high public demand for these species by people who participate in wildlife viewing as recreation. Thousands of recreational dollars are spent yearly by wildlife watchers looking for these species, particularly birds and mammals. You only need to spend 1 day of the May long weekend in the Okanagan Valley visiting "hot spots" for Okanagan "specialties" such as canyon wren (*Catherpes mexicanus*), yellow-breasted chat (*Icteria virens*), white-headed woodpecker (*Picoides albolarvatus*), and sage thrasher to realize how many recreational hours these "peripheral" species provide. After "being in the outdoors," the second most important reason British Columbian wildlife viewers give for deciding where to watch wildlife is "the variety of wildlife available" (Reid 1998).
9. Peripheral populations are likely to contain genetic stock that will make them valuable not only for conservation, but for agricultural or horticultural purposes as well. They often are populated by individuals with increased hardiness or other characteristics that make them valuable. O'Brien (1996) has noted that peripheral habitats can be an important source of new cultivars. For example, peripheral populations of giant chain fern (*Woodwardia fimbriata*) in British Columbia are regarded as having a high value for increased hardiness for this horticulturally valuable species.
10. Peripheral populations can be numerically significant. Intuitively one would expect that the number of individuals of a species would dwindle towards the edge of a species range, and that the importance of a peripheral population to the numbers of a species as a whole, would be small. This is often not the case. Barn owls (*Tyto alba*) in southwestern British Columbia are at the northern edge of the species' range, but can occur at very high densities relative to the species elsewhere (R.W. Campbell, Wildlife Branch, B.C. Ministry of Environment, Lands and Parks, pers. comm.). High densities of individuals in populations of species at the edge of their ranges have been noted for other species groups, such as warblers (e.g., Oliarnyk and Jones 1997, Bennett et al. 2000). As Lomolino and Channell (1995) point out, management practices that discount peripheral populations make the assumption that species at the edge of their range are found in small numbers, so sparsely distributed that they do not contribute significantly to a species' survival. This may not always be the case and should be determined rather than assumed.
11. Populations at the edges of geographic ranges may also be important in surviving long-term environmental changes, such as global warming (Hunter 1991, Quinn and Karr 1992). This is of particular importance in British Columbia, which, by at least 1 estimate, is expected to lose more species than any other jurisdiction in Canada due to global warming (Kutner and Morse 1996). Peripheral species may be those most suited to establishing themselves in the new habitats created by warming temperatures.
12. Several areas of endemism (areas with unique species) straddle the British Columbia border, including: the Rocky Mountains in eastern British Columbia, western Alberta, and northeastern Montana; the Georgia Depression/Puget Lowlands, the Cascades, and the Shuswap Highland–Okanagan Valley–Columbia River Basin in southern British Columbia and northern Washington; and the Northern Mountains and Plateau area in extreme north-central British Columbia and the southern Yukon (Douglas 1996). If arbitrary rules requiring a species to occur farther than a certain distance from a border are used by each state, territory, and province, these endemic species may not receive adequate attention.

CONCLUSION

If one looks at the entry for "peripheral" in Roget's thesaurus, you'll find a rather large entry—some of the synonyms include "secondary," "nonessential," "borderline," and "tangential." Given the implications outlined above, I would suggest that these words do not describe the issues associated with the listing of so-called peripheral species. Instead, the issues related to species at the edge of their ranges are at the core of biodiversity and conservation planning. Perhaps biologists concerned with conservation planning and biodiversity management should relegate the term "peripheral species" to the same place as the terms "decadent stand," "overmature forest," and "barren lake."

Listing is often the first step in a process that leads to inventory, recovery planning, and protection. It is important that scarce resources are not squandered on species that are not able to benefit from conservation attention. Since there are ecological, economic, evolutionary, and conservation issues associated with the management of peripheral species and populations, it would seem prudent to risk-list those that are imperilled. Careful planning should then be associated with the resources that are subsequently spent on managing, conserving, recovering, and restoring these species and populations.

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Geographic Distribution of Small Range Species and Development Patterns in the United States

R.F. Abbitt
University of Idaho
P.O. Box 44, Moscow, ID 83844 USA
abbi7807@uidaho.edu

ABSTRACT

In a world constantly expanding and eliminating native habitats, small range species are at greater risk of endangerment than those species with larger, more diverse ranges. In order to prevent these species from becoming the next generation of threatened and endangered species, it is integral to know where these species are located, as well as development trends in these same locations. This study has mapped the location of small range species of birds and butterflies on a county scale in the United States and identified those areas with the highest growth rates. Butterfly species occurring in 30 or fewer counties (1% of the total counties in the United States) were chosen as small range species of this taxon, and small range avian species were defined as those with breeding ranges of 50,000 km^2 or less. Avian breeding ranges were digitized using geographic information systems and then converted to a county presence/absence scale for comparison with butterfly species. The two taxa distributions were combined and any hot spots (counties with high numbers of both taxa) identified. Results of these analyses indicate that: 1) there is a relatively high county of occurrence correlation between the two taxa; 2) hot spots of these small range species occur in counties along the southern border of the United States including the states of California, New Mexico, Arizona, Texas, Louisiana, and Florida; and 3) Arizona, New Mexico, and Texas are among the top ten fastest growing states (based upon population). This southern trend relates directly to the results of two previous studies that identified counties with the highest numbers of currently endangered species. By identifying these similar county subsets, it is possible that current efforts to protect listed endangered species in these counties may simultaneously be protecting habitat for small range species of birds and butterflies. For areas with no current protection or preserve status these results may serve to identify priority areas for future conservation efforts.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Rare and Endangered Marine Invertebrates in British Columbia

William C. Austin
Marine Ecology Station and Khoyatan Marine Laboratory
R.R.1, Cowichan Bay, BC, VOR 1N0, Canada
baustin@island.net

ABSTRACT

Most of our knowledge and focus on rare and endangered species in British Columbia is on the large, furry or feathery animals and on leafy plants. We are just beginning to recognize that some local marine invertebrates and seaweeds may be at risk. Approximately 50 species of rarely seen invertebrates were identified for the British Columbia Conservation Data Centre (CDC). Some of these are at the edge of their range or in special habitats in British Columbia, and are common elsewhere. Others are known from only 1 or 2 sites in the province, and are rare or apparently absent in other parts of the world's seas. They occur in a variety of habitats ranging from intertidal lagoons to the depths of fjords, to offshore hydrothermal vents. Amateur naturalists and divers, as well as specialists, have played a role in identifying habitats where some of these animals occur.

Key words: disjunct populations, marine invertebrates, peripheral populations.

Assessments of rare and endangered species in British Columbia have emphasized terrestrial vertebrates and plants (e.g., Stace-Smith et al. 1980, Munro 1993, Harding and McCullum 1994). Scudder (1996) compiled and annotated a list of rare and endangered terrestrial and freshwater invertebrates in British Columbia. However, little has been published on rare and endangered marine species in the province (e.g., Farr and Bunnell 1980, Hawkes 1994, Nagorsen 1994, Peden 1994).

In 1996 and 1997 I collaborated with 6 specialists (M. Hawkes, seaweeds; A. Peden, fish; J. Watson, mammals; A. Burger, birds; N. McDaniel, underwater habitats; and J. Harper, marine geomorphology) in compiling a list of marine inventory requirements in British Columbia for the British Columbia Conservation Data Centre (CDC; Austin et al. 1997). This inventory includes lists of about 50 potentially rare and endangered species in each of 3 major categories (invertebrates, vertebrates, and seaweeds). The inventory also includes a list of 59 collections with major holdings of British Columbia coastal marine biota, and a list of 180 experts on the systematics of British Columbia marine animals and plants. In 1997–98 I worked with J. Booth and Associates under contract with the CDC to convert the list of invertebrate records into a database (Austin and Booth 1998). In the present paper I summarize the data from the invertebrate records.

METHODS

Species were intentionally selected from a broad range of invertebrate higher taxa. However, I only included readily visible and identifiable species, thus excluding groups such as nematodes and gastrotrichs. The selection drew on records from a species list (Austin 1985), communications with a range of specialists on marine groups (see acknowledgements), and a survey of collection records at the Royal British Columbia Museum. Species selected have been recorded at 5 or fewer sites in British Columbia.

RESULTS

The selected species are listed together with occurrence records in Table 1. The information sources for these records are given in Table 2. The details of the these records may be found in the database created by Austin and Booth (1998) and housed at the CDC. I have clustered the species according to known occurrences in both protected and unprotected habitat (Table 3).

DISCUSSION

Ranking of Species

Of the 52 species listed, all but 1 or 2 might fit the S1 category of the CDC. This category is defined as including species that are critically imperilled in the province because of extreme rarity (5 or fewer extant occurrences or very few

Text continued on p. 63.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One
B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1. Distribution of rare and endangered marine invertebrates.

Scientific name	Common name	BC sites[a]	Other sites[a]	Comments
Phylum PORIFERA	SPONGES			
Class CALCAREA	calcareous sponges			
Grantia sp. *aff. compressa*	flattened sac sponge	1	?	littoral cave, endangered
Class HEXACTINELLIDA	six rayed glass sponges	many[b]	0	uniquely shallow
Rhabdocalyptus dawsoni	boot sponge			+ unique bioherms
Aphrocallistes vastus	cloud sponge			
Class DEMOSPONGIAE	other sponges			
Halichondria sp. *aff. fibrosa*	thick white felt sponge	1	0	1 specimen, 70 m
Hymenamphiastra cyanocrypta	blue encrusting sponge	2	CA, OR	edge of range?
Plocamilla igzo	red crust sponge	2	CA	10 m, QCI, disjunct?
Chelonaplysilla polyraphis	purple slippery sponge	4	CPAC	littoral caves
Psammopemma new sp.	sand sponge	1	0	littoral cave, threatened
Phylum CNIDARIA	STINGING THREAD ANIMALS			
Class HYDROZOA	hydroids			
Corymorpha sp.	raspberry hydroid	3	0	1 local region, 10–30 m
Stylaster? sp.	white stubby hydrocoral	1	0?	18 m, Biosphere Reserve
Class ANTHOZOA	sea anemones, corals			
Isadella sp.	bamboo coral	2	?	238+ m, fjord, seamount
Synhalcurias sp.	tall, deep sea anemone	1	0	80 m, unique habitat?
Phylum PLATYHELMINTHES	FLATWORMS			
Class TURBELLARIA	free living flatworms			
Hoploplana californica	knobby flatworm	1	CA	1 specimen, cryptic
Phylum ECHINODERMATA	SPINY SKINNED ANIMALS			
Class CRINOIDEA	sea feather stars			
Antedon sp. *aff. petasus*	brooding feather star	2	?	2 specimens, N. fjords
Class HOLOTHUROIDEA	sea cucumbers			
Pentamera trachyplaca	fat Pentamera	5	CA	uncommon over range
Ekmania diomedeae	a white sea cucumber	0	AK	just N of B.C. border
Thyonidium kurilensis	purple tentacled sea cucumber	4	AK, WA	uncommon over range?
Class ECHINOIDEA	sea urchins & allies			
Echinarachnius parma	regular sand dollar	1?	AK, NF	extirpated or ID error?
Class ASTEROIDEA	sea stars			
Ceramaster arcticus	Arctic cookie star	5	AK	cold water, peripheral
Pisaster giganteus	giant spined sea star	1?	CA	extirpated or ID error?
Class OPHIUROIDEA	brittle stars			
Ophiopteris papillosa	brown spiny brittle star	2	CA	peripheral or disjunct
Ophioplocus esmarki	husky tan serpent star	1	CA	disjunct, live bearer
Phylum HEMICHORDATA	ACORN WORMS & ALLIES			
Class ENTEROPNEUSTA	acorn worms			
Saccoglossus sp.	orange acorn worm	5	?	lagoons, threatened
Phylum CHORDATA	VERTEBRATES & ALLIES			
Class ASCIDIACEA	sea squirts			
Eudistomia purpuropunctatum	purple lobular sea squirt	3	WA	1 local area: Clayoquot
Ritterella aequalisiphonis	sandy lobed sea squirt	2	WA/CA	insufficient sampling
Megalodicopia hians	megabyte sea squirt	2	DEEP	1 fjord in B.C.
Pyura sp. *cf. tesselata*	brown spot plated sea squirt	1	?	unique habitat, 30 m

Table 1. Continued.

Scientific name	Common name	BC sites[a]	Other sites[a]	Comments
Phylum BRACHIOPODA	LAMP SHELLS			
Class INARTICULATA	unhinged lamp shells			
Discinisca lamellosa	smooth tiny lamp shell	2	SEPAC	80+ m, small, cryptic
Class ARTICULATA	hinged lamp shells			
Platidia horni	tiny shouldered lamp shell	4	CA	30+ m, small, cryptic
Phylum MOLLUSCA	SOFT BODIED ANIMALS			
Class POLYPLACOPHORA	sea cradles			
Hanleyella oldroydi	tiny black spotted sea cradle	5	AK, CA	40+ m, small, cryptic
Class GASTROPODA	snails, sea slugs, limpets			
Lepetodrilus corrugatus	corrugated limpet	1	0	2,400 m, vent species
Calliostoma bernardi	beaded white top shell	1	1 - CA	130 m, 2 specimens, disjunct?
Arctomelon stearnsi	Alaskan volute	1	AK	210 m, peripheral
Fucaria striata	vent snail	1	0	2,400 m, vent species
Anidolyta spongotheres	limpet like yellow sea slug	2	0	100+ m, rare, 2 specimens
Okenia vancouverensis	Vancouver's okenia sea slug	2	0	20 m, cryptic
Flabellina iodinea	Spanish shawl sea slug	1	CA	conspicuous, peripheral
Flabellina sp.	snowy flabellina sea slug	2	?	harbours, introduced?
Cuthona punicea	pomegranate aeolid sea slug	1	0	on red *Corymorpha*
Cumanotus fernaldi	Fernald's sea slug	2	CA	harbours, introduced?
Class BIVALVIA	clams, mussels, scallops, etc.			
Rhamphidonta retifera	walking clam	2	CA	few sites over range
Phylum POGONOPHORA	BEARD WORMS			
Lamellibrachia barhami	seep tube worm	1	CA	restricted to seeps, vents
Phylum ANNELIDA	SEGMENTED WORMS			
Class POLYCHAETA	paddle footed worms			
Spinther ?alaskensis	grub-like sponge worm	1	AK?	2 specimens, rare
Hediste limnicola?	colourful red nereid	2	CA	introduction?, disjunct?
Lindaspio southwardorum	vent spionid	1	?	restricted to vents
Phylum ARTHROPODA	JOINT LEGGED ANIMALS			
Subphylum UNIRAMIA	INSECTS & ALLIES			
Class CHILOPODA	centipedes			
? *Lionyx hedgpethi*	intertidal centipede	1	CA	disjunct, rare?
Subphylum PYCNOGONIDA	SEA SPIDERS			
Sericosura venticola	vent sea spider	2	CPAC	restricted to vents
Subphylum CRUSTACEA	HARD SHELLED ANIMALS			
Class CIRRIPEDIA	barnacles			
Pollicipes polymerus	goose barnacle	2	0	subtidal pops., unique
Class MALACOSTRACA	crabs, shrimp and allies			
Mursia gaudichaudi	laterally spined crab	1	CA	accidental, 1 specimen
Hapalogaster grebnitzkii	fuzzy lithode crab	1	AK	peripheral
Lebbeus polaris	polar lebbeid shrimp	1	AK	peripheral
Emerita analoga	sand or mole crab	2+	CA, AK	non-breeding El Nino

[a] AK = Alaska; BC = British Columbia; CA = California; CPAC = central Pacific; DEEP = >200 m; NF = Newfoundland; OR = Oregon; SEPAC = southeast Pacific; WA = Washington.

[b] These species occur in many shallow sites in B.C. but in other juridictions they are known only from deeper water.

Table 2. Sources of information on species distribution, including published sources and unpublished personal communications and observations.

Scientific name	Information source(s)[a]
Phylum PORIFERA	
Class CALCAREA	
Grantia sp. *aff. compressa*	Austin (1985)
Class HEXACTINELLIDA	
Rhabdocalyptus dawsoni	Conway et al. (1991)
Aphrocallistes vastus	Austin et al. (1985)
Class DEMOSPONGIAE	
Halichondria sp. *aff. fibrosa*	Austin (1985)
Hymenamphiastra cyanocrypta	Austin (1985), N. McDaniel (pers. comm.)
Plocamilla igzo	Austin (1985)
Chelonaplysilla polyraphis	Austin (1985), N. McDaniel (pers. comm.)
Psammopemma new sp.	Austin (unpubl. data)
Phylum CNIDARIA	
Class HYDROZOA	
Corymorpha sp.	Millen (pers. comm.), N. McDaniel (pers. comm.)
Stylaster? sp.	Radcliffe et al. 1991, N. McDaniel (pers. comm.)
Class ANTHOZOA	
Isadella sp.	Austin (1985)
Synhalcurias sp.	Tunnicliffe (1985), D. Dunn (pers. comm.)
Phylum PLATYHELMINTHES	
Class TURBELLARIA	
Hoploplana californica	Austin (1985), S. Millen (pers. comm.)
Phylum ECHINODERMATA	
Class CRINOIDEA	
Antedon sp. *aff. petasus*	Austin (1985)
Class HOLOTHUROIDEA	
Pentamera trachyplaca	Lambert (1984, 1997), RBCM
Ekmania diomedeae	Lambert (1997)
Thyonidium kurilensis	Lambert (1997), A. Lambe (pers. comm.)
Class ECHINOIDEA	
Echinarachnius parma	Bush (1918), Clark (1925), Mortensen (1928)
Class ASTEROIDEA	
Ceramaster arcticus	Lambert (1981), N. McDaniel (pers. comm.), Austin (unpubl. data)
Pisaster giganteus	Austin (1985), Feder (1980), Fisher (1930), Can. Geol. Survey (?)
Class OPHIUROIDEA	
Ophiopteris papillosa	Austin (1970, 1985), N. McDaniel (pers. comm.)
Ophioplocus esmarki	Austin (1985), RBCM
Phylum HEMICHORDATA	
Class ENTEROPNEUSTA	
Saccoglossus sp.	Austin (1985), D. Quayle (pers. comm.), C. Cameron (pers. comm.)
Phylum CHORDATA	
Class ASCIDIACEA	
Eudistomia purpuropunctatum	Lambert (1989), Radcliffe et al. (1991), R. Harbo (pers. comm.)
Megalodicopia hians	Austin (1985), C. Monniot (pers. comm.)
Pyura sp. *cf. tesselata*	Austin (1985), Austin (1975)
Ritterella aequalisiphonis	Austin (1985)

Table 2. Continued.

Scientific name	Information source(s)[a]
Phylum BRACHIOPODA	
Class INARTICULATA	
Discinisca lamellosa	Bernard (1971, 1972)
Class ARTICULATA	
Platidia horni	Bernard (1971, 1972), Austin (unpubl. data)
Phylum MOLLUSCA	
Class POLYPLACOPHORA	
Hanleyella oldroydi	Ferreira (1979), RBCM
Class GASTROPODA	
Lepetodrilus corrugatus	McLean (1993)
Calliostoma bernardi	McLean (1994)
Arctomelon stearnsi	Rice (1973), RBCM
Fucaria striata	Waren and Bouchet (1989)
Anidolyta spongotheres	Bertsch (1980), Behrens (1991)
Okenia vancouverensis	Behrens (1991), S. Millen (pers. comm.), O'Donoghue (1921)
Flabellina iodinea	Behrens (1991), S. Millen (pers. comm.)
Flabellina sp.	Behrens (1991), S. Millen (pers. comm.), N. McDaniel (pers. comm.)
Cumanotus fernaldi	Behrens (1991), S. Millen (1983), N. McDaniel (pers. comm.)
Cuthona punicea	Behrens (1991), S. Millen (1985 and pers. comm.)
Class BIVALVIA	
Rhamphidonta retifera	Cowan (1964), Bernard (1975), R. Forsyth (pers. comm.)
Phylum POGONOPHORA	
Lamellibrachia barhami	Juniper et al. (1992), Tunnicliffe (pers. comm.)
Phylum ANNELIDA	
Class POLYCHAETA	
Spinther ?alaskensis	Austin (1985)
Hediste limnicola?	Carl (1937), V. MacDonald (pers. comm.), Austin (unpubl. data)
Lindaspio southwardorum	Blake and Maciolek (1992)
Phylum ARTHROPODA	
Subphylum UNIRAMIA	
Class CHILOPODA	
? *Lionyx hedgpethi*	Austin (1985)
Subphylum PYCNOGONIDA	
Sericosura venticola	Child (1987), Child and Segonzac (1996)
Subphylum CRUSTACEA	
Class CIRRIPEDIA	
Pollicipes polymerus	N. McDaniel (pers. comm.)
Class MALACOSTRACA	
Mursia gaudichaudi	Hart (1982), RBCM
Hapalogaster grebnitzkii	Hart (1980, 1982)
Lebbeus polaris	Green and Butler (1988), Jensen (1995)
Emerita analoga	Efford (1969), Hart (1982)

[a] RBCM = Royal British Columbia Museum collection records

Table 3. Clustering of species by habitat.

Habitat	Species
PROTECTED HABITATS	
Gwaii Haanas	*Plocamilla igzo*, *Pentamera trachyplaca*
Stubbs Island	*Corymorpha* sp., *Cuthona punicea*
Clayoquot Sound	*Stylaster* sp., *Eudistomia purpuropunctatum*
Broken Group	*Pyura cf. tesselata*, *Saccoglossus* sp., *Pentamera trachyplaca*
Race Rocks	*Ceramaster arcticus*
Juan de Fuca Ridge	*Lepetodrilus corrugatus*, *Fucaria striata*, *Lamellibrachia barhami*, *Lindaspio southwardorum*, *Sericosura venticola*,
UNPROTECTED HABITATS	
Portland Canal	*Megalodicopia hians*, *Isadella* sp., *Ceremaster arcticus*, *Antedon aff. petasus*, *Hapalogaster grebnitzkii*, *Lebbeus polaris*
Dixon Entrance	*Arctomelon stearnsi*
Hecate Strait	hexactinellid bioherms
Queen Charlotte Sound	hexactinellid bioherms
Dean Channel	*Antedon aff. petasus*
Slingsby Channel/Nakwakto Rapids	*Pollicipes polymerus* deep, *Ceremaster arcticus*
Nahwitti Bar	*Discinisca lamellosa*
Harvey Cove	*Ophioplocus esmarki*
Double Island, Esperanza	*Flabellina iodina*
Louie Creek	*Saccoglossus* sp., *Rhamphidonta retifera*
Slope off Clayoquot	Spinther ?alaskensis
Jervis Inlet	*Anidolyta spongotheres*
Diana Island	?*Lionyx hedgpethi*
Execution Rock	*Grantia* sp., *Chelonaplysilla polyraphis*, *Psammopemma* sp., *Hymenamphiastra cyanocrypta*, *Ritterella equalisiphonis*
Dixon Island	*Ophiopteris papillosa*, *Hoploplana californica*
Halibut Bank	hexactinellid bed, *Calliostoma bernardi*
Saanich Inlet	*Halichondria aff. fibrosa*, *Synhalcurias* sp., *Platidia horni*, *Hediste limnicola*
Selkirk Waterway	*Okenia vancouverensis*, *Cumanotus fernaldi*, *Flabellina* sp.
Cobb Seamount	*Isadella* sp.
Spieden Island	*Thyonidium kurilensis*

remaining individuals) or because of factor(s) that make them especially vulnerable to extirpation or extinction (Harcombe et al. 1994). However, to varying degrees, the few records for marine invertebrate populations may reflect limited observations rather than limited occurrences. Among the species considered here, the distribution of the mole crab (*Emerita analoga*) is the best documented (e.g., Efford 1969, Hart 1982). This species has stable populations along the outer sand beaches of California and southern Oregon, but populations occur only intermittently in Washington and British Columbia. These occurrences are correlated with El Niño events and a well-developed countercurrent running north along the coast. *E. analoga* larvae are in the plankton for more than 4 months. This is an unusually long period for marine invertebrate larvae and would be an adaptive advantage for long-distance larval transport by ocean currents (Thorson 1961, Efford 1969). I would classify *E. analoga* populations as SA in the CDC ranking system. This is defined as an element that is considered accidental or casual in the province: a species that does not appear on an annual basis (Harcombe et al.1994).

Potentially Disjunct Populations

Ten species in Table 1 have apparently disjunct distributions. They are recorded from British Columbia and California, but there are no published records from Oregon or Washington. For example, a small population of brittle stars (*Ophioplocus esmarki*) was found at 1 location at the entrance to Quatsino Sound, but the species is otherwise known only from central California and further south. *O. esmarki* is a live bearer, thus precluding planktonic larval transport as in *E. analoga*. However, this brittle star often inhabits the interstices of kelp holdfasts and would be a candidate for rafting on dislodged kelp holdfasts.

The Californian giant kelp (*Macrocystis pyrifera*) has been reported in Alaska, but not in British Columbia to date (Scagel et al. 1989, O'Clair et al. 1996). The holdfast of this kelp does get dislodged and individuals can subsequently float along the course of the prevailing currents. It has a holdfast of up to 1 m in height. The pneumatocysts, which provide flotation, reportedly lose their buoyancy after about a week at sea (O'Clair et al. 1996). However, kelp wrapped around floating logs provides another vehicle for rafting. Anthropogenic introductions, such as from ballast water and fouled boat hulls, might be suggested for a few disjunct populations associated with commercial harbours (e.g., *Hediste ?limnicola*, *Cuthona fernaldi*). Some deep-water species found in British Columbia fjords, such as the bamboo coral (*Isadella* sp.) and the "megabyte" sea squirt (*Megalodicopia hians*), might be isolated from deep-sea populations as a result of isostatic rebound of portions of the coastline.

Could some of these disjunct populations represent separate incipient or cryptic species? There is no compelling evidence for intertidal or shallow subtidal marine refugia over the last period of glaciation, and re-population of ice-free coastal areas would not have occurred more than 10,000–15,000 years ago (e.g., Hebda and Frederick 1990). This is a short period of time for speciation among the macro-invertebrates considered here.

Peripheral Populations

Species with populations primarily in the arctic and subarctic, but with isolated occurrences in British Columbia, are typically found in cold-water fjords such as Portland Canal at the northern end of coastal British Columbia (e.g., *Hapalogaster grebnitzkii*) or in the open waters at Dixon Entrance (e.g., *Artomelon stearnsi*). However, others also occur farther south in particularly cold waters such as at Race Rocks (e.g., *Ceramaster arcticus*). The apparently disjunct occurrences of the typically southern forms noted above would be better classified as peripheral populations if intervening records are found from Oregon and Washington.

Species Apparently Limited to British Columbia

Some species are known only from within British Columbia. These include *Okenia vancouverensis*, *Stylaster* sp., *Anidolyta spongotheras*, *Corymorpha* sp., *Cuthona punicea*, *Pyura cf. tesselata*, and *Psammopemma* sp. In addition, others are limited to what are perceived as unusual or isolated habitats such as a periodically anoxic fjord (*Halichondria aff. fibrosa*, *Synhalcurius* sp.), or restricted to a single hydrothermal vent field (*Lepetodrilus corrugatus*, *Fucuria striata*). It is perhaps premature to characterize these as endemic to British Columbia, given our limited field surveys. However, based on our present knowledge, their rare occurrence within the province coupled with the absence of records from outside the province is cause for concern.

Common Species in Unusual Habitats

The goose barnacle (*Pollicipes polymerus*) is a common intertidal species on wave-exposed coasts in the northeast Pacific. The subtidal occurrence of a population at Nakwakto Narrows, where tidal currents reach 16 knots, is apparently unique for this species. Similarly, the very shallow water (5 m) occurrence of hexactinellid sponges in British Columbia is apparently unique in the world, as is their occurrence in bioherms (biological reef mounds) up to 10 m in height.

Are There any Extirpated Marine Invertebrate Species in the Province?

Many invertebrate species could have disappeared from the province unnoticed, but I could find only 2 documented assessments. The sand dollar (*Echinarachnius parma*), primarily an arctic species, is recorded by Mortensen (1928) as occurring on this coast from Alaska to Vancouver Island and Puget Sound. Clark (1925) identified a specimen in the

British Museum that was labeled from Puget Sound. I can find no subsequent records from British Columbia or Puget Sound. The sea star (*Pisaster giganteus*) was reported from Saanich Inlet by de Loriol (1897) and from Vancouver Island by Verrill (1909). Fisher (1930) suggests that the figure by de Loriol appears to be *P. ochraceus*, a common species in British Columbia. Although the distribution of *P. giganteus* is often cited as extending into British Columbia (e.g., Feder 1980), I can find no collecting records since those noted above. All of these authors are experts on echinoderms, and the characters distinguishing *E. parma* and *P. giganteus* from other locally occurring species are readily observable.

CONCLUSIONS

1. Most of the listed invertebrate species are known from 5 or fewer provincial records and would be considered extremely vulnerable based on the criteria set by the CDC. However, we cannot exclude the possibility that their apparent rarity is, in part, a reflection of limited field surveys.
2. The species listed are not necessarily the most rare or endangered in British Columbia. Many additional marine invertebrate species in the province are known from 5 or fewer records.
3. Based on limited field data, some species are at the periphery of their range in the province; others have populations in British Columbia, but are otherwise known only from central California south; still others are recorded only within the province.
4. Two species found early in this century in British Columbia have not been recorded since then in the province, although they are common elsewhere within their range. However, extirpation and extinction of marine invertebrates are difficult to substantiate (e.g., Carlton et al. 1991, Glynn and De Weerdt 1991, Glynn and Feingold 1992).
5. Approximately 30% of the listed species are found in regions that are designated or officially proposed as protected habitats. In most cases it is not clear what this protection entails, other than discouraging physical removal of benthic species.
6. While it may be argued that, on a global scale, marine extinction rates are lower than those on land, there is clearly a need to address the relatively neglected issue of local extirpation in the marine environment (e.g., Norse 1993, McKenney 1998).

RECOMMENDATIONS

1. Confirm the status of the listed species in the province based on additional existing information by: assessing additional museum material in private hands and outside the province; questioning amateur naturalists and divers; and soliciting reviews by systematic specialists.
2. Visit known species population sites to assess: population sizes; reproductive status; other appropriate aspects of biology; and proximal and more general potential anthropogenic impacts.
3. Increase public awareness of populations at risk through presentations, publications, and Web site displays.
4. Implement appropriate protection for populations at risk.
5. Monitor populations and their habitats.

ACKNOWLEDGEMENTS

I thank the many people who suggested either candidates for status as rare and endangered species or habitats containing these species. These people include: M. Arai, C. Cameron, R. Forsyth, G. Green, G. Jenson, A. Lambe, P. Lambert, N. McDaniel, J. McLean, S. Millen, V. Tunnicliffe, S. Cannings, J. Harper, K. Simmington, and M. Zacharias. J. Booth was instrumental in developing the database and entering some of the data. K. Sendall helped in searching for museum material, and A. Harcombe and L. Ramsay provided guidelines and suggestions on assessment of rare and endangered species status.

The following institutions provided facilities for study of material in museums or in the field: Bamfield Marine Station; Royal British Columbia Museum; Institute of Ocean Sciences; Pacific Biological Station; and University of Victoria. The study was supported by the CDC, and the Corporate Resource Inventory Initiative. Thanks, too, to the CDC for supporting the publication of this paper.

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Freshwater Molluscs at Risk in British Columbia: Three Examples of "Risk"

Jacqueline S. Lee
Natural Resources and Environmental Studies
University of Northern British Columbia
Prince George, BC, V2N 4Z9, Canada

Josef Daniel Ackerman
Natural Resources and Environmental Studies
University of Northern British Columbia
Prince George, BC, V2N 4Z9, Canada
ackerman@unbc.ca

ABSTRACT

Thirteen families of freshwater molluscs, representing both gastropods (snails) and bivalves (mussels and clams), are known to occur in British Columbia. Freshwater molluscs may be at risk due to a variety of reasons including: 1) habitat loss/alteration; 2) restricted niche requirements; 3) confused taxonomy; and 4) a general lack of information on their distribution and ecology. Specifically, these factors may: 1) interfere with the ability of freshwater molluscs to reproduce; 2) impact molluscs directly or indirectly through the introduction of exotic species; and 3) change the amount of habitat that is available to molluscs. We present results for 3 species of snails that were assessed to be at risk in British Columbia. *Physella wrighti* (hotwater physa) is at risk due to restricted niche requirements, while the assessment of risk for *P. hordacea* (grain physa) may be in error due to confused taxonomy within the genus. A lack of information on the distribution of freshwater molluscs led to the incorrect assessment of *Acroloxus coloradensis* (Rocky Mountain capshell) to be at risk. The concept of risk may have many interpretations, but careful laboratory and field studies have led to realistic designations.

Key words: *Acroloxus coloradensis*, freshwater limpet, freshwater molluscs, habitat degradation, hotsprings snail, *Physella hordacea*, *Physella wrighti*.

Freshwater molluscs are the snails, mussels, and clams that inhabit inland waters. These organisms are components of all aquatic ecosystems and occur in almost every type of freshwater habitat, from large lakes and rivers to small, even vernal, ponds (Pennak 1989). Bivalves (i.e., mussels and clams) are generally suspension feeders, while gastropods (i.e., snails) are primarily surface grazers. As such, molluscs are primary consumers in aquatic ecosystems (Brown 1991, McMahon 1991), and consequently they have been used as environmental indicators (McMahon 1991, Reynoldson et al. 1995). Freshwater molluscs are also prey items for ecologically and economically important aquatic and terrestrial predators (Pennak 1989).

Current taxonomic information indicates that 13 of the 15 families of freshwater molluscs known to occur in Canada are represented in British Columbia by 74 species (Table 1; Clarke 1981). Recent field and museum research indicates that there are additional species records for British Columbia, and that the ranges described for many species are in need of revision (Lee and Ackerman unpubl. data). Beyond species records, there is limited information available as to the ecological and environmental factors that may have led to the present distribution of freshwater molluscs in British Columbia. This lack of knowledge has put freshwater molluscs at risk because conservation requires scientific data as a basis for action.

Table 1. Freshwater mollusc families of Canada (Clarke 1981). The "√" indicates the presence of members of the family in British Columbia.

Class Gastropods (snails)	Class Pelecypoda (mussels and clams)
Subclass Prosobranchia (gilled snails)	
Order Mesogastropods	Order Eulamellibranchia
√ (Family Viviparidae	√ Family Margaritiferidae
√ (Family Valvatidae	√ Family Unionidae
√ (Family Hydrobiidae	√ Family Sphaeriidae
Family Truncatellidae	
Family Bithyniidae	
√ Family Pleuroceridae	
Subclass Pulmonata (lunged snails)	
Order Basommotophora	
√ Family Acroloxidae	
√ Family Lancidae	
√ Family Lymnaeidae	
√ Family Physidae	
√ Family Planorbidae	
√ Family Ancylidae	

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 2. Three freshwater mollusc species assessed by COSEWIC (Committee on the Status of Endangered Wildlife in Canada) as At Risk or Potentially At Risk in British Columbia.

Scientific name	Common name	Original collection		
		Site	Collector(s)	Date
Physella wrighti	hotwater physa	Alpha Stream, Liard River Hotsprings	A.H. Clarke and D. Wright	19 Aug 1973
Physella hordacea	grain physa	(1) Long Lake, Nanaimo, Vancouver Island	A.H. Clarke and B.T. Kidd	31 Jul 1972
		(2) Magic Lake, North Pender Island	MacDougall	15 Jan 1979
Acroloxus coloradensis	Rocky Mountain capshell	Purden Lake, 60 km E of Prince George	A.H. Clarke and D. Wright	8 Aug 1973

From the perspective of conservation biology, the designation of "at risk" can result from a number of causes. For example, a given species may be at risk because: 1) its habitat is being altered or lost; 2) it has restricted niche requirements; 3) its taxonomic status is confused; and/or 4) its distribution and ecology are not well known.

Freshwater molluscs are potentially at risk because of the impacts of human activities, which alter aquatic ecosystems and habitats. There are at least 3 ways that freshwater molluscs can be impacted by human activities:

Reproduction

Aquatic organisms living in flowing systems must ensure that their reproductive efforts are not lost downstream. Freshwater mussels in North America have overcome this problem by evolving glochidia larvae that are parasitic on fish (McMahon 1991). After the larvae contact a fish host, they are encysted by fish tissue, where they develop and undergo metamorphosis. After 10–30 days, the juvenile mussel breaks out of the cyst and falls to the bottom in a habitat that may be far upstream from that of its parent. Any process that prohibits the contact of the mussels with their fish hosts (e.g., dam construction or impoundment), or that degrades habitat such that fish hosts are lost, puts freshwater molluscs at risk. This is, in effect, an indirect impact that may limit the reproductive success of the mussels through the removal of fish hosts (Williams et al. 1993).

Exotic Species Introductions

Exotic species introductions can directly or indirectly affect freshwater molluscs. Indirect effects, such as the introduction of exotic fish species, can result in the loss of native fish species that are the hosts of the glochidial larvae. Direct impacts result from introduced species, such as zebra mussels (*Dreissena* spp.). Zebra mussels attach to the shells of native mussels interfering with feeding, respiration, balance, burrowing, and locomotion, and also reduce or remove food resources from the water column (Nalepa and Schloesser 1993, Parker et al. 1998). Interestingly, zebra mussels have free-swimming veliger larvae, a condition that was conserved evolutionarily, and which has facilitated their rapid expansion in North America (Ackerman et al. 1994).

Habitat Loss/Alteration

Aquatic habitats can be lost when wetlands are filled in for development, and when meandering rivers are channelized (e.g., Okanagan River, B.C.; Cannings and Cannings 1996). The alteration of waterfront for recreation, housing, or commercial development can decrease the diversity of habitats available for molluscs. While most landscape alterations are detrimental to freshwater mollusc habitat, a study conducted in the Torpy River watershed in east-central British Columbia indicates that the building of access roads to facilitate timber harvest can augment or create aquatic habitat for the freshwater clam, *Pisidium casertanum* (Lee and Ackerman unpubl. data). Thus, in some special cases, landscape alteration may not have a negative effect on freshwater mollusc habitat. Clearly, our knowledge of the ecology of freshwater molluscs needs improvement.

The present knowledge of the British Columbia fauna of freshwater molluscs led Scudder (1996) to list 14 species of potentially rare or endangered freshwater molluscs in British Columbia. Of these, 3 species, all snails, (*Physella wrighti* [hotwater physa], *P. hordacea* [grain physa], and *Acroloxus coloradensis* [Rocky Mountain capshell]), had only been collected in 1 locality within the province (Table 2). Therefore, these species were selected by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) for further assessment. These 3 species are used in this paper as examples of species at risk.

METHODS

A search of museum records was conducted to ensure that all species location records had been incorporated correctly in Scudder (1996). This search revealed a second location record for *P. hordacea* at Magic Lake on North Pender Island (see Table 2). The first stage of our field research was to

attempt to locate the populations at these sites.

Physella wrighti

(Fig. 1) This species appears to be endemic to the Liard River hotsprings complex in northern British Columbia (Te and Clarke 1985). The population of *P. wrighti* was located by following Alpha Stream downstream from its exit out of Alpha Pool in Liard River Hotsprings Provincial Park in 1997. *P. wrighti* was identified by its occurrence in the type locality. The snails were found living within mats of the green alga, *Chara*, on the surface of the soft sediments and on birch leaves (*Betula papyrifera*) that had fallen into the stream. The population size was estimated by pushing an 18-cm-diameter stainless steel sieve (nominal pore size 1.5 mm) under the *Chara* in each area and shaking the algae to dislodge snails. Where the snails occurred on open sediment the numbers were counted within 400-cm^2 quadrats.

Physella hordacea

(Fig. 2a) (1) Long Lake: Three trips were made to Long Lake in 1996 and 1997. Specimens from the family Physidae were located on the undersides of lily-pads (*Nuphar* sp.) in 2 areas of the lake, and were collected by hand. As snail densities were relatively low in this lake, only about one-half of the snails observed were collected in order to minimize impacts to the population. (2) Magic Lake: A collection of physids was made from a park at the west end of Magic Lake in 1998. Here, the physids were abundant and were readily collected by hand from the undersides of lily-pads and by sweeping near-shore vegetation with a stainless-steel sieve.

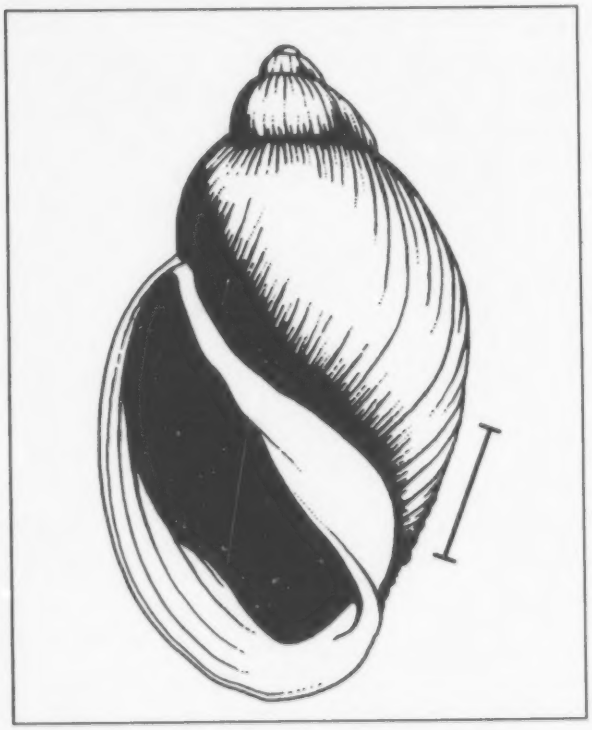

Figure 1. *Physella wrighti* (Te and Clarke 1985). Length of line equals 1 mm.

Acroloxus coloradensis

(Fig. 3) The population of *A. coloradensis* in Purden Lake was located in the summer of 1997. The entire perimeter of Purden Lake was surveyed during the summer. A study of the freshwater molluscs of northern British Columbia revealed populations of *A. coloradensis* at 6 additional sites (Lee and Ackerman 1999*a*). This species is easily recognized by its unique, prominent and extended, acutely pointed apex of the shell, located posteriorly and on the left (Lea 1864). *A. coloradensis* is small and adheres closely to the substrate. Specimens were found by close examination of substrates removed from the water.

RESULTS

Physella wrighti

The entire population of *P. wrighti* is in a 34 X 2 m reach of Alpha Stream. The population estimate made within this reach found 979 snails, which likely represents approximately one-half of the population (i.e., estimated population of about 1,735 snails, based on sampling uncertainty on the

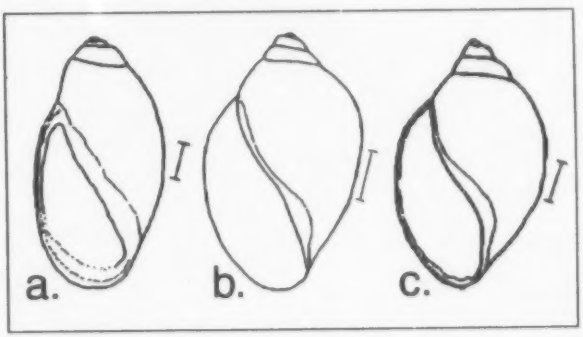

Figure 2. Drawings of Physids: (a) *Physella hordacea* (Lea 1864) from holotype drawing by Dr. S. Wu; (b) Specimen collected at Long Lake, B.C. by Clarke and Kidd, 1972, identified by Te (as per museum label) as *Physella hordacea* and by Wu (pers. comm. 1998) as *Physella* (*Physella*) *heterostropha*; (c) Specimen collected at Magic Lake, B.C. by MacDougall, 1979, and identified by Smith (as per museum label) as *Physella hordacea* and by Wu (pers. comm. 1998) as *Physella* (*Physodon*) *integra*. Length of line equals 1 mm.

Table 3. Museum holdings of *Physella hordacea*. (CMN = Canadian Museum of Nature; MCZ = Museum of Comparative Zoology; UMMZ = University of Michigan Museum of Zoology; USNM = United States National Museum; ANSP = Academy of Natural Sciences; n.d. = no date.)

Location	Museum holding and date
BRITISH COLUMBIA	
Long Lake, Nanaimo	(CMN ML067486 - 1972)
Magic Lake, North Pender Island	(CMN ML 086048 - 1979)
Vancouver Island	(MCZ 55263 - 1924; UMMZ 193163 - n.d.)
WASHINGTON	
Columbia River at Vancouver	(CMN ML 055081; USNM 170764 - n.d., but as holotype/paratype, collected in 1864 or earlier)
Vancouver Island	(UMMZ 124596 - n.d.; UMMZ 193163 - n.d.)
OREGON	
Vancouver Island	(ANSP 17231 - n.d.; MCZ 302538 - 1924)
Empire City	(USNM 29208 - n.d.)
Small stream entering Wilmot River, 6 miles above Portland	(ANSP 112435 - 1912/1913)
Oregon	(USNM 99307 - n.d.; ANSP 123965 - prior to 1882; UMMZ 115664 - n.d.)

different substrate types). The full report of the current status of *P. wrighti* is available in Lee and Ackerman (1999*b*).

Physella hordacea

This species appears to be restricted to British Columbia, Washington, and Oregon (Burch 1989). A summary of known museum holdings is given in Table 3. The information supplied on the locations has been somewhat problematic. The holotype and paratype localities should concur, but the holotype locality description (Lea 1864) is "Vancouver Island, Oregon," while the paratype location description is "Columbia River at Vancouver, Washington." Clarke's (1981) publication of this species in Long Lake, B.C., as well as the uncertainty about the precise type locality has led some authors to believe that Long Lake may be the type locality.

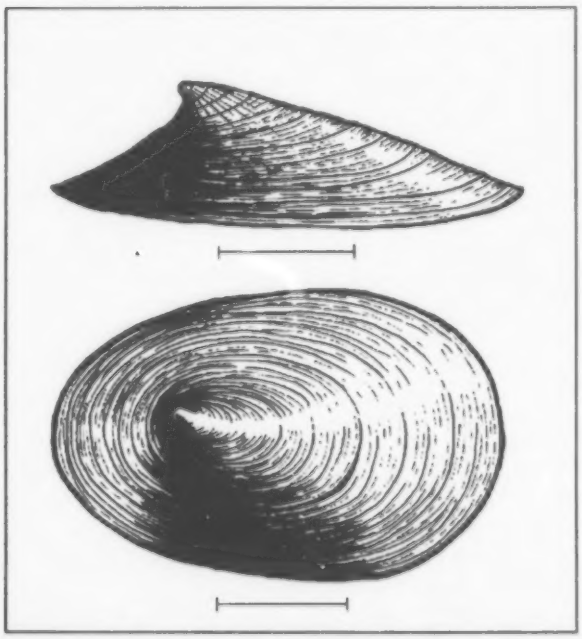

Figure 3. *Acroloxus coloradensis* (J. Henderson 1930). Length of line equals 1 mm.

The average population density for physids in Long Lake was calculated to be 4.3×10^{-5} physids/m^2 of surface area, based on the size of the lake (i.e., 4.2×10^5 m^2 surface area). Thus, we estimate a population size of 20 physid snails in the entire lake. Clarke (1972) expended 1.33 person-hours of search effort at Long Lake and his collection in the Canadian Museum of Nature contains >50 specimens of physids. The maximum number of physids observed during our 1.5 person-hours of search effort was 16. This suggests that in 1972 physids were much more common in Long Lake than they were during the current study. No population estimate was made at Magic Lake.

Acroloxus coloradensis

This species displays a seemingly rare and disjunct distribution (Bryce 1970). In Canada, there are 9 published records of collections of *A. coloradensis* (2 in Alberta [Mozley 1930]; 5 in Ontario and Quebec [Clarke 1970]; 1 in British Columbia [Clarke 1981]; and 1 in Alberta [Paul and Clifford 1991]).

During the surveys of Purden Lake, the variability in the observed density of *A. coloradensis* and the narrow range of depths that could be examined from the shore precluded the possibility of making an accurate population estimate. However, there appears to be a large (i.e., at least several hundred) and nonlocalized population of *A. coloradensis*

Table 4. Locations, substrate, and protection for *Acroloxus coloradensis* sites in British Columbia.

Location	Substrate	Protection
Purden Lake	undersides of rocks in shallow water	Westernmost portion of lake is a provincial park. *A. coloradensis* has been collected from waters adjacent to the park.
Shane Lake	submerged wood and debris such as fiberglass	Adjacent land owned by provincial government and leased to city of Prince George. It is managed as a park by Forests for the World (a non-government agency).
Carp Lake	submerged wood and rocks	Entire lake is in Carp Lake Provincial Park.
Gataiga Lake	submerged wood	Lake is in a B.C. Ministry of Forests Recreation Site.
Babine Lake pond	submerged deciduous leaves	Pond is entirely within Red Bluffs Provincial Park.
Topley rest area pond	decaying outside leaves of cattails (*Typha latifolia*)	North shore of pond was accessed from Ministry of Highways rest area. Status of remainder of area adjacent to pond is not known.
Decker Lake	rocks and submerged wood	The only area searched was adjacent to private property.

in Purden Lake. The 7 known sites in British Columbia for *A. coloradensis*, the substrate types from which it was collected, and the protection of these sites is given in Table 4. With the report of an additional site for *A. coloradensis*, in a lake northeast of Edmonton, Alberta (T. Hoover, University of Northern British Columbia, December 1998, pers. comm.), there are now 16 known sites for *A. coloradensis* in Canada (Fig. 4).

DISCUSSION

Physella wrighti

It is difficult to say why the snails inhabit such a limited area of the stream, but they have been found in no other locations within this hotsprings system, and no other snail populations were located at other hotsprings surveyed in the area (Te and Clarke 1985). Given that this single population lives

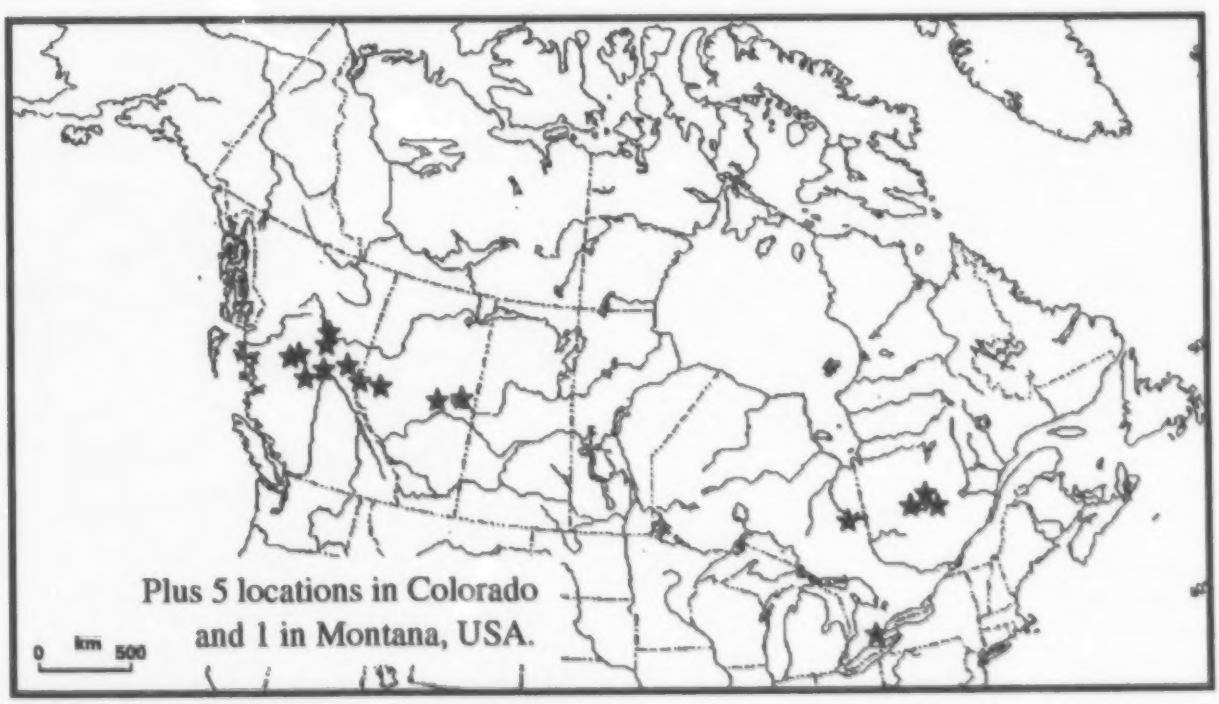

Figure 4. Approximate locations of 22 known sites for *Acroloxus coloradensis* in North America.

downstream of a pool estimated to be used by 100,000 bathers per year (W. Woodhouse, British Columbia Parks, 1997, pers. comm.), this species is at risk. The largest threat to these snails would likely come from the inadvertent addition of substances (e.g., soap, shampoo, oil) to the pool, from a sudden change in the flow if the outlet to Alpha Stream were blocked, or from direct human contact. Recently, bathers who had illegally climbed a fence in Banff National Park to swim in a hotspring pool caused the death of an estimated 100 *P. johnsoni*, a species listed as Threatened by COSEWIC (Bachusky 1999). Due to the possibility that the entire population of these snails could be eliminated by a single event, *P. wrighti* has been listed by COSEWIC as Endangered. Potential conservation efforts may benefit from the observation that we have cultured *P. wrighti* ex situ for >1.5 years.

The status of *P. wrighti* is an example of where our current knowledge of British Columbia's freshwater mollusc fauna has yielded a unique species, which is only found in a unique habitat. In this case, the species is at risk because of restricted niche requirements.

Physella hordacea

Based on the examination of specimens and photographs sent to him, Dr. Shi-Kuei Wu (Professor of Natural History, University of Colorado, Boulder, CO) is of the opinion that the collection of physids made at Long Lake in 1972 were identified incorrectly. He has identified the physid collections from Long Lake as *Physa* (*Physella*) *heterostropha*. (Different authors use different generic names within the family Physidae, in that *Physa* or *Physella* can be used to refer to the same species. The use can depend on the date of the publication and/or the taxonomic preference of the author. We use the nomenclature convention of the particular scientist cited.). Wu has also re-identified the Magic Lake collections as *Physa* (*Physodon*) *integra* and supplied a copy of his drawing of the holotype for *P. hordacea* (Fig. 2a).

Using conformity to holotype shell morphology as an identifying characteristic, it is evident that Figures 2b and 2c are different from Figure 2a. Figures 2b and 2c appear to be similar, but Wu has concluded that these 2 collections are actually from different subgeneric groups because of important differences in internal structures. Based on Wu's re-identifications and our own assessment of comparative shell morphology, it does seem that the Long Lake and Magic Lake specimens may have been identified incorrectly. The type specimens may indeed be from the Oregon/Washington area as indicated on their collection labels.

Our current assessment of the status of *P. hordacea* (Lee and Ackerman submitted *a*), which is under review by COSEWIC, is that until such time as there is more conclusive evidence of its occurrence in Canada, *P. hordacea* should be classified as Indeterminate, as *P. hordacea* is a species for which there is insufficient scientific information to support status designation. Notwithstanding this recommendation, there is still concern for the population of physid species in Long Lake. This is an example of where confused taxonomy has led to the assessment of a species to be potentially at risk.

Acroloxus coloradensis

The evidence suggests that *A. coloradensis* has a more widespread distribution and is adapted to a much wider range of habitats than was described previously. There are now 22 known locations for *A. coloradensis*, 16 in Canada and 6 in the United States (U.S. Department of the Interior 1994; Fig. 4). Our current assessment of the status of *A. coloradensis* (Lee and Ackerman submitted *b*), which is under review by COSEWIC, is that *A. coloradensis* be considered Not at Risk. This is an example of where the lack of information on the distribution and ecology of British Columbia's freshwater molluscs has led to the incorrect assessment of a species to be potentially at risk.

MANAGEMENT IMPLICATIONS

Factors that put freshwater molluscs at risk in British Columbia fall into several categories, all of which pose current threats to the native fauna of British Columbia. Assessment of the immediate risk to particular species, such as those reviewed above, is hampered by the lack of information on the distribution and ecology of British Columbia's freshwater molluscs. As freshwater molluscs are important components of British Columbia's inland waters, management requires a better understanding of this group of organisms for the maintenance of these essential ecosystems.

ACKNOWLEDGEMENTS

We wish to thank the members of the Physical Ecology Laboratory at the University of Northern British Columbia for their assistance with this project. Particular thanks is due to L. Wilson for her assistance in the field. Funding for these studies was supplied in part by COSEWIC, the British Columbia Conservation Data Centre, and Forest Renewal British Columbia.

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Tracking Rare Native Vascular Plants in British Columbia

Jenifer L. Penny
British Columbia Ministry of Environment, Lands and Parks,
Conservation Data Centre
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada
Jenifer.Penny@gems3.gov.bc.ca

George W. Douglas
British Columbia Ministry of Environment, Lands and Parks,
Conservation Data Centre
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada

ABSTRACT

The British Columbia Conservation Data Centre (CDC) is currently tracking >600 rare native vascular plant taxa in the province. This overwhelming number, about 25% of our entire native flora, requires careful prioritization to maximize the efforts of a limited staff and a restricted budget. The highest priority goes to the 241 Red-listed plants, whose habitats are either threatened or endangered. Little is known about the abundance and condition of populations of these critical taxa, and inventory is therefore much needed. In recent years, intensive field surveys have vastly increased the knowledge and occurrences both for critical areas and specific taxa. In many cases, the CDC database has increased tenfold for these areas and taxa. This inventory has also enabled a number of taxa to be de-listed.

Key words: abundance, inventory, populations, rare plants, tracking, vascular plants.

The British Columbia Conservation Data Centre (CDC) is currently tracking >600 rare native vascular plant taxa in the province, which is about 25% of our entire native flora. Rare plants are categorized[1] by the CDC as Red-listed or Blue-listed, according to their abundance, the quality and condition of their populations, the potential threats to their survival, and whether or not they are present in protected areas in British Columbia. Once this information is known for each taxon, efforts can be prioritized. There are 241 Red-listed rare plants and 359 Blue-listed rare plants. The Red List contains the threatened or endangered plant taxa, the high priority taxa, whereas the Blue List contains those taxa that are vulnerable. Our knowledge to date was derived from a thorough survey of the major herbaria in Canada, pertinent literature, and accounts from field botanists. Ongoing inventory and participation from field workers is constantly needed to update our tracking list and keep it accurate. Less than half of the taxa on the tracking list have been adequately surveyed, and there has been little participation outside of the CDC and its contractors. Knowing this, and the fact that little research has been done on native vascular plant taxa in British Columbia, it is difficult to accurately describe the state of the rare flora.

The status of the most critical taxa (Red List) is better known than those on the Blue List. The Red-listed taxa are each known from ≤6 sites in the province, and usually occur in areas of heavy resource use or urban development, whereas the Blue-listed taxa are known from ≤21, but >6, sites, and are not endangered. Inventory efforts since the CDC was founded have been focused on the critical taxa, but have also included some vulnerable taxa as well. These studies have resulted in great advances in what is known about the taxa's conservation status. In some cases, a plant that was once thought to be Red-listed turned out to fall in the vulnerable category following intensive inventory. For example, Scouler's corydalis (*Corydalis scouleri*) was once thought to be endangered, but will now be considered vulnerable due to the findings from an extensive inventory in the Nitinat and Klanawa river valleys and the compilation of a status report. Other examples include the down-listing of the great chain fern (*Woodwardia fimbriata*), the coastal wood fern (*Dryopteris arguta*), and Howell's montia (*Montia howellii*), all rare plants from the Gulf Islands.

There have also been a number of vulnerable taxa that have recently been de-listed as a result of field work; these are Nuttall's quillwort (*Isoetes nuttallii*), tiny mousetail (*Myosurus minimus*), Newcombe's butterweed (*Sinosenecio newcombei*), and Ussurian water-milfoil (*Myriophyllum ussuriense*). These taxa are now known from >21 sites, most of which are not in danger of being destroyed. De-listing these species gives us the opportunity to focus on the truly rare plants. *Sinosenecio newcombei*, although completely

[1] The CDC is part of an international conservation network, The Nature Conservancy of the United States, which has a highly organized set of procedures for ranking rare elements and prioritizing conservation efforts. In addition to the Red and Blue lists, B.C. Ministry of Environment, Lands and Parks designations, and state and global ranks are also assigned to the rare taxa in compliance with The Nature Conservancy procedures.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 450pp.

restricted or endemic to the Queen Charlotte Islands, is so common on the islands that it no longer fits the necessary criteria to be tracked as a rare plant. There are no apparent threats to the 57 known occurrences. There are also likely many more occurrences than we already know about on the islands. Unfortunately, the CDC is too limited in staff and resources to track plants that fall short of the Blue List but are regionally important.

Despite all the de-listing, there are still 241 endangered or threatened plants to consider. Some of the rarest plants on the tracking list are only known from 1 site; there are 103 of these on the list. In these cases, there are 2 possibilities: either the plant is extremely rare and endangered, in which case its existence is extremely precarious; or it has been poorly surveyed and a few more sites may eventually be discovered. The southern maidenhair fern (*Adiantum capillus-veneris*) occurs in only 1 site, likely in part due to its habitat specificity; it only occurs at a hotsprings seepage at Fairmount Hotsprings. There are other hotsprings localities in the province, so it may also be that its dispersal ability is limited. Other plants that are only known from 1 site include buff daisy (*Erigeron ochroleucus* var. *scribneri*), western wahoo (*Euonoymus occidentalis*), and dwarf hesperochiron (*Hesperochiron pumilus*). The former was collected for the first time in British Columbia in the Elk River valley in extreme eastern British Columbia by Hans Roemer in 1996, and will probably be found in more sites in the future. Both *E. occidentalis*, known only from Courtenay, and *H. pumilus*, which has only been found near Salmo, have been documented for several years, and no more sites have been discovered. It is less certain that they could be found in more sites.

The tracking list statistics indicate that there is a problem. Why are there so many threatened and endangered plants in British Columbia? Furthermore, what is being done about it? Development pressure and the altering of habitats are the biggest threats, which mainly apply on private lands. Therefore, protected areas created from Crown lands miss the mark for rare plants in British Columbia. Regional parks have also been problematic. Other solutions are needed.

One of the reasons for these overwhelming statistics is that there has been a great deal of habitat loss over the past century. There are several species in the development "hot spots" that have a number of sites reported from the turn of the century, but only exist in a few or none of these at this time. Development "hot spots" include southeastern Vancouver Island, greater Vancouver (including the whole Fraser valley), the south Okanagan, and some areas in the Kootenays. Other major threats in general include grazing, introduced species, agriculture, and other forms of habitat alteration. There are 4 taxa that have been extirpated and 30 taxa that are historic (last observed before 1950) in British Columbia. Historic taxa may be extirpated or may have merely evaded collectors for decades. Historic taxa are generally found in heavily developed areas or habitats that have been extremely altered, thus lending support to their being extirpated. The statistics are not reassuring.

There has been a general lack of awareness of the plight of rare species and lack of planning for their persistence in areas of heavy urbanization demands. Regional parks especially have fallen short in providing adequate protection. A population of prairie lupine (*Lupinus lepidus* var. *lepidus*) was destroyed in Beacon Hill Park in Victoria in 1994 when workers removing old posts from along the road bank unknowingly wiped out its habitat. A population of coast microseris (*Microseris bigelovii*), a plant in the sunflower family, was also destroyed in a regional park when a bench was placed over top of it.

Another example of the lack of procedure in place for sound management in regional parks occurred in Uplands Park, Victoria. A fire truck drove into the middle of the park to service a fire hydrant in winter making tire ruts in a large vernal pool, which is home to a host of rare plants; after a complaint was made, they decided to fill in the ruts with gravel, which was quite destructive to its inhabitants.

Therefore, it is clear that some parks do not afford much protection for rare plant populations. Strict guidelines regarding proper management for species and their habitats need to be prepared and utilized. A major component of this is communicating the management strategies to all levels within the organization, from the director down to the people doing work in the parks. Part of the procedure also has to involve consultation with the CDC or knowledgeable individuals or organizations.

This lack of awareness and action to eliminate threats to rare taxa exists not only at the regional level, but also within the provincial government, which has had, and continues to have, protected areas strategies and our program, the CDC. However, we still have too many rare taxa and little protection for them. Only one-quarter of the Red-listed taxa occur in protected areas in the province, and only 10% of their populations fall in these protected areas.

Although inventory is still needed, it not a lack of information (which is housed in the CDC and available to anyone that asks for it) that accounts for the overwhelming number of rare plants. Nor is it due to the fact that there is no protected areas strategy in the province, because there are programs in place. The problem lies in the fact that most occurrences of rare taxa in the province are found on private land, as opposed to lands owned by the Crown. Therefore, 1 of the biggest concerns overall is the lack of endangered species legislation, which could address rare plants on private lands and their habitats.

We are now nearing the year 2000 and neither British Columbia nor Canada have any rare species legislation in place. The Canadian bill that has been proposed, still has not passed and would only protect those taxa on federal lands in

British Columbia. This greatly reduces the ability of this bill to effectively address the threats to our rare and endangered plants in the province. Legislation at the provincial level would be much more desirable. Ontario and Manitoba have rare species legislation that protects plants. In Ontario, plants must be listed by the Committee on the Status of Endangered Species in Canada (COSEWIC) in order to be protected, and they may occur on either private or public lands. Although this is a very political process that takes quite a lot of time to see through to the end, it is still a very strong act. Likewise, Arizona has had rare species legislation since 1929. In addition, the United States as a whole has a strong endangered species act.

Without rare species legislation there is not much hope for rare plant taxa in British Columbia. The efforts of groups such as The Land Conservancy, The Nature Trust, and the Nature Conservancy of Canada are the best tools that we currently have to protect rare plants on private land. However, purchases by these groups to date have only affected a small percentage of our rare plant occurrences. Land stewardship programs and covenants, which are usually associated with legislation, although not in British Columbia, would be another effective tool. Covenants to protect rare plants do exist, but are not common in the province. Therefore, there is currently no strong program in place to adequately protect rare plants. There is still much work to do to convince the policy-makers that legislation is the only tool that could adequately protect rare plants and that it is a worthwhile cause. Ultimately it means protecting biodiversity in the province of British Columbia.

Habitat Use and Distribution of Listed Neotropical Migrant Songbirds in Northeastern British Columbia

Stephen Bennett
Mirkwood Ecological Consultants Ltd.
501 - 10th Avenue, Castlegar, BC, V1N 1K6, Canada
stbennet@wkpowerlink.com

Peter Sherrington
R.R.#2, Cochrane, AB, T0L 0W0, Canada

Pierre Johnstone
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 2958, Fort Nelson, BC, V0C 1R0
Canada

Bruce Harrison
409 - 9215 94A Street
Fort St. John, BC, V1J 6E4, Canada

ABSTRACT

The primary objective of this paper is to summarize the distribution and habitat use of Red- and Blue-listed neotropical migrant songbirds (hereafter referred to as "listed" species) in the Fort Nelson Forest District (FNFD) in northeastern British Columbia. Data were summarized from 2 main sources: anecdotal data and studies conducted prior to 1996; and 4 Forest Renewal British Columbia (FRBC) studies conducted since 1996. We also compared the relative density of listed species (pairs/ha) to densities reported in the central and eastern parts of their breeding range. The dominant biogeoclimatic zone in the FNFD is the Boreal White and Black Spruce (BWBS) zone (68% of the district), which represents a large part of the mixedwood boreal forest found in British Columbia. Northeastern British Columbia is home to 6 listed species that are rare or unrecorded in the rest of the province: bay-breasted warbler (BBWA), black-throated green warbler (BTNW), Canada warbler (CAWA), Cape May warbler (CMWA), Connecticut warbler (COWA), and Philadelphia vireo (PHVI). The listed species are of particular management concern because they have a relatively limited distribution in British Columbia, they use habitats often targeted for forest development, and little is known about their distribution and breeding ecology in the province. All the studies in the FNFD found listed species using similar habitats to those described in the literature. BTNW and COWA were rare and are likely restricted to the southern portion of the FNFD. PHVI also appeared to be rare, but were sporadically distributed throughout the eastern portion of the FNFD and as far north as southeastern Yukon. BBWA, CAWA, and CMWA were all common in their preferred habitats. The densities of common listed species (BBWA, CAWA, CMWA) and several other neotropical migrants (magnolia warblers, ovenbirds, Tennessee warblers) were similar to densities reported from the central and eastern parts of their ranges. This suggests that the FNFD has optimal habitat for these listed species and others, and that western portions of the boreal forest may have unique ecological characteristics that respond to human development differently than eastern portions of the boreal forest.

Key words: boreal forest, distribution, habitat use, neotropical migrants, northeastern British Columbia, songbirds.

The boreal region is the only forested circumpolar terrestrial biome, and includes very large forested wilderness areas interspersed with very large peatland wilderness areas, covering approximately 1.6 billion ha (Pastor et al. 1998). Today, the boreal forest is developed over significant portions of previous ranges, largely as a result of forest harvesting (Niemi et al. 1998). In British Columbia, the Boreal White and Black Spruce biogeoclimatic zone (BWBS) exemplifies the lower elevation component of the boreal region. The BWBS zone occurs throughout northern British Columbia, but is most common in the Fort Nelson Forest District (FNFD) in the northeastern portion of the province. Much of the BWBS falls in the "enhanced resource development zone" (LRMP 1997) and as such, has a lower biodiversity emphasis option (B.C. Ministry of Forests and B.C. Ministry of Environment Lands, Lands and Parks 1995).

A number of songbirds that are rare or unrecorded in the rest of the province breed in the northeastern portion of British Columbia (Enns and Siddle 1996), including 6 that are listed as either Red or Blue species by the Conservation Data Centre (CDC) of the British Columbia Ministry of

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Environment, Lands and Parks (MELP), and are hereafter referred to as "listed" species; Red-listed species are considered to be endangered or threatened in British Columbia, and Blue-listed species are considered to be vulnerable or sensitive (B.C. Ministry of Environment Lands, Lands and Parks 1998). Red-listed species believed to breed in the FNFD include the Cape May warbler (*Dendroica tigrina*; CMWA), the bay-breasted warbler (*Dendroica castanea*; BBWA), the Connecticut warbler (*Oporornis agilis*; COWA), and the black-throated green warbler (*Dendroica virens*; BTNW); Blue-listed species include the Canada warbler (*Wilsonia canadensis*; CAWA) and the Philadelphia vireo (*Vireo philadelphicus*; PHVI; Ministry of Environment Lands, Lands and Parks 1998). Very few nesting sites have been confirmed in British Columbia and information is limited regarding the distribution and abundance of these species (Campbell et al. 1997, 1999). Although their habitat requirements have been described in eastern North America, it is still unclear how applicable the descriptions are to northeastern British Columbia.

These species are of particular management concern because they are all neotropical migrants, meaning they breed in temperate regions but migrate long distances from their wintering grounds in the tropical Americas. Recent studies have indicated a widespread decline in populations of neotropical migrants, and provincial status reports on these species suggest that there could be significant negative impacts from forest development (Cooper et al. 1997*a-f*).

There are 2 principal natural resource development industries in the FNFD: forest harvesting, and oil and gas development. The forest industry has had a significant and growing impact since the early 1970s. Original development targeted the valley-bottom white spruce (*Picea glauca*) stands and, as a result, these stands are relatively rare. The rate of harvest is likely to increase, particularly in the upland deciduous and mixedwood stands (M. Thorp, British Columbia Ministry of Forests, Fort Nelson, 1999, pers. comm.).

The natural gas industry has been active in the FNFD since the 1950s (LRMP 1997). Operations begin with seismic exploration, typically straight lines cut with a bulldozer, which may be hundreds of kilometres long. Seismic activity is increasingly laid out in a grid pattern over hundreds of square kilometres. The result is a bewildering pattern of intersecting straight lines, over the entire district. Once a likely gas source has been pinpointed, a well site is cleared, typically 1.5–5 ha, and a drilling rig is installed. Together, exploration and development may have cleared an average of 1,500 ha per year since the 1970s, or approximately 57,000 ha to date (M. Thorp, pers. comm.).

The objectives of this paper are to summarize FNFD bird inventory results, summarize the distribution and habitat use of listed forest-dwelling neotropical migrants (BBWA, BTNW, CAWA, CMWA, COWA, PHVI), comment on the relative abundance of species in this area compared to the centre of their range, comment on possible impacts from forestry and mining, and suggest future research required.

STUDY AREA

The FNFD is bounded to the east by Alberta, to the north by the Yukon and the Northwest Territories, to the west by the northern Rocky Mountains, and to the south by the Fort St. John and Mackenzie forest districts (Fig. 1). It is the second largest forest district in British Columbia, covering approximately 8.2 million ha. The BWBS is the dominant biogeoclimatic zone, found over 68% of the district. This landscape features long, very cold winters, where monthly average temperatures are <0°C for half the year (Meidinger and Pojar 1991). The major tree species are white and black spruce (*Picea mariana*), lodgepole pine (*Pinus contorta*), tamarack (*Larix larcina*), balsam fir (*Abies balsamea*), paper birch (*Betula papyrifera*), balsam poplar (*Populus balsamifera*), and trembling aspen (*Populus tremuloides*). Forest fires are the dominant natural disturbance, though frequent outbreaks of defoliating insects also occur (B.C. Ministry of Forests and B.C. Ministry of Environment Lands, Lands and Parks 1995). The district has relatively high forest productivity compared to other neighbouring boreal forest districts, due to the FNFD's high topographic complexity, extensive riparian forest habitat associated with large rivers, coarse-textured and well-drained soils, high precipitation, and high latitude (giving long summer daylight hours; D. Patterson, Forest Management Division, Alberta Department of Environmental Protection, 1999, pers. comm.). The Boreal Plains, Northern Boreal Mountains, and Taiga Plains ecoprovinces are represented in the study areas. For a complete description see Demarchi (1990).

METHODS

Since 1996, MELP has initiated 4 bird inventories in the FNFD to gain a better understanding of boreal forest bird communities. These studies have been funded by Forest Renewal British Columbia (FRBC). For more details on survey methods please refer to the specific reports (Bennett and Enns 1996, Bennett et al. 1998, Savignac 1998, Bennett 1999). The Smith River, the Dunedin airstrip (an area accessed by the Dunedin airstrip between Torpid Creek and the Dunedin River), Dunedin River (valley bottom of Dunedin River, accessed by helicopter west of the Dunedin airstrip), and Prophet River were surveyed in 1998. In 1997, 261 northern goshawk (*Accipiter gentilis*) call-playback sites scattered throughout the west-central portion of the FNFD were sampled (with a 5-min listening period for songbirds), and in 1996 the Liard River near the "Big Bend" was surveyed.

We generated distribution maps for each listed species as in Enns and Siddle (1996), with locations from recent surveys

(Bennett and Enns 1996, Bennett et al. 1998, Bennett 1999) plotted over the previous ranges as reported by Godfrey (1986) and Enns and Siddle (1996). We summarized the habitat use for each listed species from recent studies in the FNFD (Bennett and Enns 1996, Enns and Siddle 1996, Bennett et al. 1998, Savignac 1998, Bennett 1999), and common habitat attributes are presented in tabular fashion. The habitats that listed species used in the FNFD were compared to the habitats they used in other regions of the boreal forest. Densities of listed species were estimated from point count and transect data from the FRBC studies and compared to density estimates in the literature.

Figure 1. Location of Fort Nelson Forest District and study areas in northeastern British Columbia.

RESULTS

General

The number of neotropical migrant songbirds[1] recorded throughout the FNFD appeared to decrease the farther north and west studies were conducted in the FNFD. The Prophet River study site in the southern portion of the FNFD had the most neotropical migrants (27 species; Savignac 1998), and the Smith River in the northwest portion of the FNFD had the least (20 species; Bennett et al. 1999).

Since 1996, 357 individual records of listed species have been made in the FNFD. These records account for approximately 80% of all records of listed species collected in the FNFD to date (S. Cannings, British Columbia Conservation Data Centre, 1999, pers. comm.). The new records also represent predominantly new sites where the species have been recorded. Previous records were generally along the Alaska Highway, at provincial parks, and near the Fort Nelson airport. The Prophet River was the only site where all 6 listed species were recorded in the FNFD, and it is the only site where BTNW, COWA, and PHVI were recorded. BBWA, CAWA, and CMWA were all recorded at the Liard, northern goshawk, and Dunedin sites, whereas the CMWA was the only listed species recorded at the Smith River site.

All the studies found species using habitats similar to those described in the literature, with the exception of the CAWA. Almost all of the records of CAWA since 1996 were in mature aspen-dominated forests, whereas in studies outside the FNFD, CAWA were frequently recorded in younger, cut-over areas (McLaren 1987). Mature and old forests appeared to be preferred by all listed species except the COWA and to a lesser extent PHVI, which used pole-stage aspen (Savignac 1998). All listed species were generally restricted to valley-bottom sites and mid-slopes along major rivers, except CMWA, which were found in mid to upper elevations throughout the FNFD (Bennett and Enns 1996, Bennett 1999).

There is a unique assemblage of western montane and eastern boreal forest bird species in the FNFD because it is bordered on the west by the Rocky Mountains and the east by the northern Alberta Plain. "Western" species such as western tanager (*Piranga ludoviciana*), Townsend's warbler (*Dendroica townsendi*), varied thrush (*Ixoreus naevius*), and Hammond's flycatcher (*Epidonax oberholseri*) were common and recorded in the same locations as "eastern" species such as BBWA, magnolia warbler (*Dendroica magnolia*; MAWA), ovenbird (*Seiurus aurocapillus*; OVEN), and red-breasted grosbeak (*Pheucticus ludovicianus*; Bennett and Enns 1996, Bennett et al. 1998, Greenberg and Sterling 1994, Savignac 1998, Bennett 1999). The FNFD is also uniquely situated at the southern extent of many arctic breeding species and the northern extent of many southern breeding species. For example, we found evidence of breeding Pacific loons (*Gavia pacifica*) and arctic terns (*Sterna paradisaea*) in the same area as breeding pairs of black terns (*Chlidonias niger*) and soras (*Porzana carolina*).

Bay-breasted Warbler

Abundance/Population Status

BBWA were relatively common in their preferred habitat. The density of BBWA was the third highest out of 20 species in mature mixed riparian forest and fifth highest out of 31 species in mature coniferous riparian forest at the Liard River (Bennett and Enns 1996). Eastern North American populations have been demonstrated to fluctuate widely depending on spruce budworm (*Choristoneura fumiferana*) levels (Williams 1996). No long-term studies have confirmed that this phenomenon occurs in the western part of their range. A North American-wide decline in BBWA populations has not been demonstrated, although spraying of vast areas of forest to control for spruce budworms, and forest harvesting are anticipated to cause population declines. Their status is listed as follows: Provincial listing - Red; CDC provincial rank - S2B, SNZ; CDC global rank - G5; COSEWIC (Committee on the Status of Endangered Wildlife in Canada) - not listed; British Columbia Identified Wildlife - no.

Distribution

BBWA breed across the boreal forests of Canada and northern New England with the bulk of the population thought to be in Ontario, Quebec, and the Maritime provinces (Dunn and Garrett 1997). The breeding range extends from east-central and northeastern British Columbia to extreme southeastern Yukon, and across northeastern Alberta as far as western Newfoundland (Dunn and Garrett 1997). BBWA winter from Costa Rica and Panama to northwest South America east of the Andes (Williams 1996, Dunn and Garrett 1997).

The breeding distribution has been confirmed in the Yukon, as a BBWA nest site was found in the lower La Biche River drainage in 1996 (Eckert 1998). Enns and Siddle (1996) also recorded secondary evidence of breeding northwest of Fort Nelson in 1992. Records since 1996 have expanded the range west by approximately 75 km. No nest sites have been located in British Columbia, but males have been recorded defending territories, and males and females have been seen carrying food to suspected nest sites (Bennett and Enns 1996).

Habitat Use

Enns and Siddle (1996) proposed a BBWA habitat model of mixedwood stands dominated by tall, closed-canopy, large spruce, with aspen and birch also common in the canopy and

[1] We used the list of neotropical migrants used by Savignac (1998) to compare total numbers between studies. Only forest songbirds were considered (i.e., swallows, shorebirds, and other open-habitat species are not included in the totals presented).

a tall (1–2 m) shrubby understory of highbush cranberry (*Viburnum edule*), red-osier dogwood (*Cornus stolonifera*), and green alder (*Alnus crispa*). All studies since then have found BBWA using similar habitat, with the exception of balsam poplar also being associated with BBWA sites (Bennett and Enns 1996, Bennett 1999). It appears that the farther north and west BBWA are found in the FNFD, the more restricted they are to valley-bottom and mid-slope mature mixed forests. In the south-central portions of the FNFD, the BBWA has been found at numerous upland-slope sites and does not appear to be restricted to large river valleys (Bennett et al. 1998, Savignac 1998). However, at the "Big Bend" on the Liard River and in the Dunedin River, BBWA were restricted to valley-bottom and mid-slope sites (Bennett and Enns 1996, Bennett 1999).

Black-throated Green Warbler

Abundance/Population Status

The BTNW is likely a rare breeding bird in the FNFD. Both studies that recorded the BTNW in the FNFD were in the southern portion of the district and only 9 records were made (MacNaughton 1995, Savignac 1998). BTNW are found in higher numbers in the Fort St. John and Dawson Creek forest districts (Merkens and Booth 1998; C. Siddle, Vernon, 1999, pers. comm.). Long-term declines and range contractions have occurred in Wisconsin, Michigan, and southern Ontario, but North American trends are confused by year to year population fluctuations (Dunn and Garrett 1997). The greatest abundance of BTNW are thought to occur in Maine, Nova Scotia, and New Brunswick. Their status is listed as follows: Provincial listing - Red; CDC provincial rank - S2B, SNZ; CDC global rank - G5; COSEWIC - not listed; British Columbia Identified Wildlife - no.

Distribution

BTNW breed across the boreal forests of Canada from east-central British Columbia and northern and central Alberta to Newfoundland (Dunn and Garrett 1997). The BTNW also breeds in central Minnesota and the Appalachian Mountains (Dunn and Garrett 1997). BTNW winter mainly in northeast Mexico south to Central America.

The breeding distribution has been confirmed by MacNaughton (1995), Siddle (1992) and Enns and Siddle (1996) who, combined, have >122 records of the BTNW in the Boreal Plains and Taiga Plains ecoprovinces around Fort St. John. Records since 1996 have expanded the range north by approximately 100 km. No nest sites have been located in British Columbia (Campbell et al. 1999).

Habitat Use

Enns and Siddle (1996) proposed a BTNW habitat model of tall mixedwood stands of spruce and poplar with mid-aged, mid-canopy aspen (BWBS balsam poplar riparian habitat type). They also found BTNW in the same habitat used by OVEN. The stands tended to be mesic with relatively shrubby understories of highbush cranberry, prickly rose (*Rosa acicularis*), bunchberry (*Corus canadensis*), and fireweed (*Epilobium angustifolium*). The only study to find BTNW since 1996 was the Prophet River study. At the Prophet River, BTNW were found in a relatively closed, mature mixed stand and a relatively open, aspen-dominated stand with a thick shrub layer (Savignac 1998). It appears that the BTNW is more restricted to old, valley-bottom forests than the other listed warblers, as it is rarely found in upper-slope sites (Savignac 1998; C. Siddle, pers. comm.).

Canada Warbler

Abundance/Population Status

CAWA were relatively common in most of the surveys and often were 1 of the most common species in their preferred habitat. For example, at the Liard River study site, CAWA were the most abundant species in mature deciduous upper-slope forest. At the Prophet River, CAWA were only recorded twice, but all habitat types were not systematically surveyed (Savignac 1998). Sharp population declines have been noted in Pennsylvania due to fragmentation; however, it appears to be stable over much of its range (Dunn and Garrett 1997). Their status is listed as follows: Provincial listing - Blue; CDC provincial rank - S3, S4B; CDC global ranking - G5; COSEWIC - not listed; British Columbia Identified Wildlife - no.

Distribution

CAWA breed across the boreal forests of Canada, from north-eastern British Columbia to Nova Scotia and south to central Alberta across to northeastern Ohio (Dunn and Garret 1997). They winter almost exclusively in South America east of the Andes in Columbia, Peru, Ecuador, and Venezuela (Dunn and Garret 1997).

The breeding distribution has been confirmed in British Columbia by Siddle (1992), Enns and Siddle (1996), Bennett and Enns (1996), Bennett et al. (1997), Savignac (1998) and Bennett (1999), who have recorded 141 CAWA in the Boreal Plains and Taiga Plains ecoprovinces between Fort St. John and the Yukon–British Columbia border. Records since 1996 have expanded the range west by approximately 50 km. No nest sites have been located, but evidence of breeding has been observed (i.e., males and females carrying food; males defending territories). CAWA have also been recorded frequently in the southeastern portion of the Yukon along the Liard River (C. Machtans, Western Arctic Habitat Program, Canadian Wildlife Service, 1998, pers. comm.).

Habitat Use

Enns and Siddle (1996) proposed a CAWA habitat model of steep unstable slopes with thick undergrowth, often dominated by soopolallie (*Shepherdia canadensis*) and rose. Birch was the

dominant tree cover, and spruce, aspen, and balsam poplar were also present. Windthrown trees and debris were common. Almost all records of CAWA since 1996 have been in mature aspen stands with closed canopies, open midstories (tree stems spaced widely), and dense understories often dominated by alder and highbush cranberry (Bennett and Enns 1996, Bennett et al. 1998, Savignac 1998, Bennett 1999). Two records along the Liard River were from young deciduous forest types (Bennett and Enns 1996). Other literature suggests that CAWA prefer riparian habitats and shrubs along streams (Dunn and Garrett 1997), but most of our records have been in upper-slope forests. These records suggest that CAWA require dense shrub habitat but can exploit both mature and young forest types depending on location and availability.

Cape May Warbler

Abundance/Population Status

CMWA were relatively common in most of the surveys and often were 1 of the most common species in their preferred habitat. For example, CMWA were the fourth most common bird recorded in the mature conifer habitat at the Dunedin airstrip, and the seventh most common in old coniferous habitat at the Smith River. Population status estimates are problematic for CMWA populations because they fluctuate more than most warblers, due to their reliance on spruce budworm populations (which are known to be very cyclic). Some local decreases and increases have been noted with no clear North American-wide trend (Dunn and Garrett 1997, Baltz and Latta 1998). The cyclic nature of CMWA populations has only been confirmed in the eastern part of its range. Their status is listed as follows: Provincial listing - Red; CDC provincial rank - S2B, SNZ; CDC global rank - G5; COSEWIC - not listed; British Columbia Identified Wildlife - no.

Distribution

CMWA breed across the boreal forests from southeastern Yukon and northeastern British Columbia across Canada to southwestern Newfoundland (Dunn and Garrett 1997) and are generally thought to be more common in the eastern part of their range (Dunn and Garrett 1997). CMWA winter primarily in the Bahamas and islands in the western and southern Caribbean (Baltz and Latta 1998).

The breeding distribution has been confirmed in the southeastern Yukon, where CMWA were recorded along the La Biche and Beaver rivers (C. Machtans, pers. comm.). Their distribution was also confirmed in northeastern British Columbia, where they were recorded around Fort Nelson (Enns and Siddle 1996). Records since 1996 have expanded the range northwest by approximately 150 km. No nest sites have been located, but males have been recorded defending territories, and males and females have been seen carrying food to suspected nest sites (Bennett and Enns 1996, Bennett et al. 1998, Bennett 1999).

Habitat Use

CMWA occupy mature to medium-aged coniferous forests, or bogs dominated by black and/or white spruce (Dunn and Garrett 1997, Baltz and Latta 1998). Enns and Siddle (1996) proposed a CMWA habitat model of white spruce in relatively dense, tall stands on flat ground with an open mossy understory. Almost all records of CMWA since 1996 have been in mature black and/or white spruce stands with open moss-dominated understories (Bennett and Enns 1996, Bennett et al. 1998, Savignac 1998, Bennett 1999). CMWA were found in young (40–80 yr old) to old (>140 yr old) forests, but most of the sightings were in mature forest. Tall spruce trees protruding from the canopy were often used by males for singing perches.

Connecticut Warbler

Abundance/Population Status

The COWA is not a common species anywhere in its range (Dunn and Garrett 1997). There is some evidence that it has declined since the nineteenth century, but it is unclear why, since breeding and wintering habitat appears to be relatively secure (Cooper et al. 1997*f*, Dunn and Garrett 1997). Their status is listed as follows: Provincial listing - Red; CDC provincial rank - S2B, SNZ; CDC global ranking - G4; COSEWIC - not listed, British Columbia Identified Wildlife - no.

Distribution

COWA breed across the boreal forests in Canada in a relatively narrow band from northeastern and east-central British Columbia and central Alberta, to central Quebec (Dunn and Garrett 1997). It is generally uncommon and local over much of its range (Dunn and Garrett 1997). COWA wintering range is poorly understood, but they are thought to winter entirely in South America in the Amazon region.

The breeding distribution has been confirmed by Siddle (1992) and Enns and Siddle (1996), who recorded 26 COWA around Fort St. John. COWA are more abundant in the Dawson Creek and Fort St. John forest districts (Merkens and Booth 1998; C. Siddle, pers. comm.). Since 1996, COWA have only been recorded along the Prophet River in the FNFD, which is a range expansion north by approximately 100 km (Savignac 1998). No nest sites have been located, but males have been recorded defending territories, and males and females have been seen carrying food to suspected nest sites (Campbell et al. 1999).

Habitat Use

Habitat varies widely throughout its range from wet conifer bogs to well-drained deciduous woodlands (Dunn and Garret 1997). Enns and Siddle (1996) proposed a COWA habitat model of mature, well-spaced, aspen-dominated forests on flat to gently sloping terrain with intermittent white spruce. The understory was well developed with a noticeable gap between the shrub layer and the canopy layer. Since 1996,

COWA have only been recorded at the Prophet River (Savignac 1998). All records of COWA at the Prophet River were in immature aspen stands (Savignac 1998).

Philadelphia Vireo

Abundance/Population Status

PHVI appear to be relatively rare and locally distributed, and their distribution is likely associated with the presence of adequate deciduous forests (Campbell et al. 1997). They may be vulnerable to logging of northeastern deciduous forests, but their ability to use second-growth forests likely makes them less vulnerable than other northeastern songbirds (Cooper et al. 1997*b*). Nonsignificant increases and decreases in populations have been noted in Canada and the United States, and their habitat appears to be relatively secure (Moskoff and Robinson 1996). Their status is listed as follows: Provincial listing - Blue; CDC provincial rank - S3S4B; CDC global ranking - G5; COSEWIC - not listed; British Columbia Identified Wildlife - no.

Distribution

PHVI breed across the boreal forests of Canada from southeastern Yukon and northeastern British Columbia to southwestern Newfoundland (Godfrey 1986, Campbell et al. 1997). PHVI winter from Guatemala south to central Panama (Ehrlich et al.1988).

Breeding of PHVI has been confirmed by Enns and Siddle (1996), who recorded 6 PHVI around Fort St. John and Fort Nelson, and the Canadian Wildlife Service, which recorded PHVI in southeastern Yukon (C. Machtans, pers. comm.). Since 1996, PHVI have only been recorded at the Prophet River (Savignac 1998), which is within their previously defined range (Enns and Siddle 1996). Four nest sites have been located in stands of trembling aspen and 1 in a balsam poplar stand (Campbell et al. 1997).

Habitat Use

PHVI typically use tall, young aspen or balsam poplar stands with 80–100% canopy closure (Campbell et al. 1997). Some mixed aspen and spruce forests are also used and both old-growth and second-growth stands are used, but second-growth stands are probably preferred (Campbell et al. 1997). Enns and Siddle (1996) propose a PHVI habitat model of dense, rapidly growing young aspen (approximately 20 years old), with lots of branches in the mid to lower canopy, and a 80–100% canopy closure. Associated understory species included Indian paintbrush (*Castilleja miniata*), fireweed, highbush cranberry, alder, willow (*Salix* spp.), and clover (*Trifolium hybridum*). Since 1996, PHVI have only been recorded at the Prophet River (Savignac 1998). All records of PHVI at the Prophet River were at the edge of old forests and old clearcuts dominated by deciduous trees (Savignac 1998).

DISCUSSION

Status and Distribution

CMWA, CAWA, and BBWA appear to be widely distributed in the FNFD and common in their preferred habitat. The CMWA probably has the widest distribution throughout the district and within individual valleys, as it utilizes valley bottom to upper elevations more readily than BBWA and CAWA. BTNW and COWA are rare in the district and appear to have a much more restricted and patchy distribution. They are mainly restricted to the southern and eastern portions of the district. PHVI appear to have a similar distribution to BTNW and COWA, as they are rare in the FNFD and sporadically distributed. However, PHVI have been recorded in southeastern Yukon along the La Biche River, which enters the Liard River approximately 25 km northwest of the "Big Bend" study site (C. Machtans, pers. comm.). This suggests that PHVI occur throughout the FNFD at least as far east as Fort Nelson and north to the Yukon border. As Campbell et al. (1997) noted, PHVI are restricted to areas with adequate young to mature deciduous forests. The songs of the PHVI and red-eyed vireos (*Vireo olivaceus*; REVI) are very similar, and the PHVI may even mimic the REVI, which also makes it difficult to determine the true distribution of PHVI.

It is unknown whether populations of any of the listed species are declining throughout North America or specifically in British Columbia because there are limited data and a high degree of variability in population levels from year to year, and ranges contract and expand periodically (Cooper et al. 1997*a-f*). However, all the listed species utilize young to mature forest habitats to some degree and these habitats are being harvested at an increasing rate in the FNFD. Based on the past 3 years of inventory work, a review of the status of listed neotropical migrants in northeastern British Columbia is necessary to update the current designations. We believe BBWA and CAWA are likely still at risk because they appear to utilize lower elevation habitats along major river systems that have already been harvested extensively. BTNW, COWA, and PHVI are also likely still at risk because of their rarity and restricted, patchy distribution in the district. CMWA appear to be the least at risk because they are more widely distributed throughout the FNFD.

Abundance and Habitat Use

Many of the species found in the study areas, including all the Red- and Blue-listed species, have a breeding distribution that extends from the hardwood forests of eastern North America through the boreal forest, to terminate in northwestern British Columbia and the southern Yukon (Godfrey 1986, Campbell et al. 1997, Dunn and Garrett 1997). The designation of these species as "eastern" has given rise to the idea that the breeding heartland of such species is in the eastern provinces of Canada, with numbers

diminishing westwards until finally a scattering of isolated pairs are found at the western terminus of the range. The listed status of a number of these species in northeastern British Columbia, together with published range maps that show them barely reaching the province, appear to have supported this contention. To test this idea, 6 neotropical migrant warbler species were selected and their breeding densities in the present study areas were compared to published breeding densities elsewhere in Canada and the northeastern United States. Three of these species, (CMWA, BBWA and Tennessee warbler [*Vermivora peregrina*; TEWA]), are spruce budworm-specialists. The other three, (CAWA, MAWA, and OVEN), are more generalist insectivores, which exploit a wide range of invertebrate food and have populations that tend to be stable over long periods. The breeding ranges of these 6 species terminate in northern British Columbia or adjacent southern Yukon, with the exception of TEWA, which extends across most of British Columbia to the west coast (Dunn and Garrett 1997).

Each of these warbler species appears to have surprisingly high breeding densities in suitable habitat in northeastern British Columbia, even, in some cases, beyond the previously known western limit of the species distribution (Figs. 2 and 3). In general, the densities are comparable to, or higher than, those in the central and eastern parts of their ranges. As most individuals of these species migrate to the area from the east or southeast they must travel considerably greater distances than their eastern counterparts to reach their breeding grounds. Rather than being considered marginal to their breeding range, it appears that much of the habitat in northeastern British Columbia is optimal for their breeding requirements and birds may be being rewarded for the extra expenditure of energy on migration by higher breeding success. For a more complete discussion of these comparisons see our unpublished report (Bennett et al. 1999).

CONCLUSION

Studies in the FNFD have focused primarily on reconnaissance-level inventories with a secondary objective of developing habitat models for listed species. Accurate distribution maps will take many years to complete, as access is severely limited during the summer months. However, habitat suitability maps should be created for all listed species for the entire Fort Nelson Forest District based on recently proposed habitat associations. Areas mapped as highly suitable should be deferred from harvesting whenever possible until a review of the status of northeastern warblers is completed. Detailed nest searches should also be conducted later in the season (end of June, first half of July) to confirm successful breeding of listed species and to compare nest productivity in the FNFD with eastern populations.

The habitat used by listed species in the FNFD was similar to those reported in the literature, with BBWA, BTNW, CAWA, and CMWA using mature deciduous to coniferous-dominated forests, and COWA and PHVI using young to mature deciduous-dominated forests. Densities of common listed species (BBWA, CAWA, CMWA) and several other neotropical migrants are similar to densities reported from the eastern and central parts of their range. This suggests that the FNFD has optimal habitat for listed species and others. The FNFD also has a unique assemblage of "eastern" species and "western" species characteristic of the western cordillera, and marks the southern and/or northern limit of several species; such avifauna and their habitat should be protected. Further management recommendations and suggestions for future research are described in the full report.

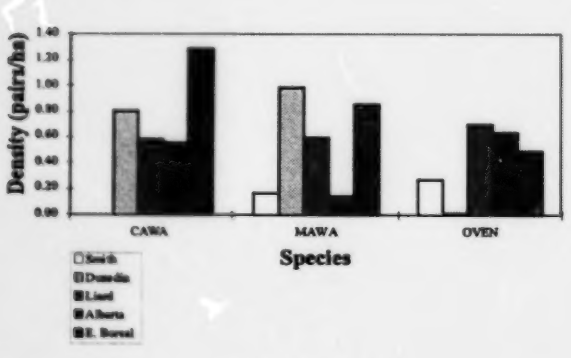

Figure 2. Comparison of densities of budworm specialists (BBWA = bay-breasted warbler, CMWA = Cape May warbler, TEWA = Tennessee warbler).

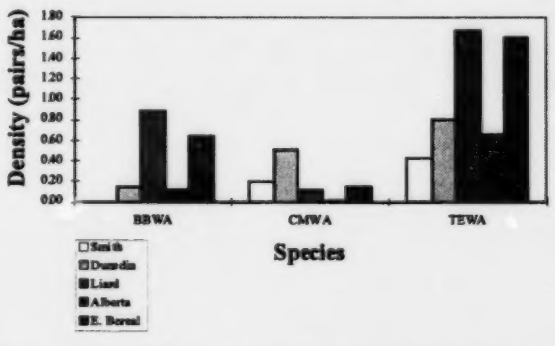

Figure 3. Comparison of densities of non–budworm specialists (CAWA = Canada warbler, MAWA = magnolia warbler, OVEN = ovenbird).

ACKNOWLEDGEMENTS

We would like to thank L. Wilkinson of the British Columbia Ministry of Environment, Lands and Parks in Fort St. John for securing funding for these projects, reviewing the papers, and providing mapping and technical support throughout the projects. M. Gill, C. Machtans, W. Nixon, and P. Sinclair of the Canadian Wildlife Service, and C. Eckert of Canadian Parks also provided reports, data, and insightful comments. Timberland Consultants and Pandion Ecological Research of Nelson, B.C. provided information on the Prophet River bird inventory. C. Savignac was the author and field biologist for the Prophet River study. The staff of the British Columbia Ministry of Forests and Slocan Forest Products in Fort Nelson also provided much-needed resources and local knowledge, which increased our efficiency in the field.

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Dragonflies at Risk in British Columbia

Leah R. Ramsay
British Columbia Conservation Data Centre
Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada
Leah.Ramsay@gems4.env.gov.bc.ca

Sydney G. Cannings
British Columbia Conservation Data Centre
Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada

ABSTRACT

In 1993, British Columbia's dragonflies and damselflies (Order Odonata) were listed and ranked as to their conservation status in the province, based upon the knowledge at the time. Inventory efforts were then focused on the species considered at risk, in order to more accurately determine their status and to identify specific sites for conservation. For 3 years, surveys were conducted in 5 regions of the province: the northeast corner, the Lower Mainland, southern Vancouver Island, the Okanagan, and the Columbia Basin. During the course of these surveys, known ranges of many species were extended, knowledge of habitat requirements increased, and 3 new species were confirmed for the province. While many of the targeted species were found only sparsely or not at all, a number of them were discovered to be more abundant than previously thought, and their conservation ranks were changed accordingly.

Key words: dragonfly, inventory, Odonata.

One of the goals of the British Columbia Conservation Data Centre is to portray as accurately as possible the status of species and habitats in the province. The first stage in this process is to draw up initial lists of all of the species of the province, with each species given a numerical value ranking its rarity or endangerment. The ranks are based on a number of factors, including the number of populations, the number of individuals, population trends, threats, and distribution. Each species is assigned an "S" (provincial) rank, ranging from 1 (most at risk) to 5 (not at all at risk; Harcombe 1994).

The dragonflies and damselflies (Order Odonata) are a relatively well known group of insects that are present in a wide variety of streams, lakes, and wetlands. In British Columbia these wetland habitats have been and continue to be altered, lost, or destroyed due to development, agriculture, and resource extraction (Stevens et al. 1995). Dragonflies are considered a priority group for inventory (Scudder 1996) because, unlike most invertebrates, they can be relatively easily identified, even in the field in many cases. They are upper-level predators in the invertebrate food-chain and have often been identified as indicators of ecosystem health (Walker and Corbet 1975, Carle 1979, Trevino 1997).

In 1993, all of British Columbia's odonates (82 at the time) were ranked by the Conservation Data Centre as to their known conservation status. However, despite the fact that the Odonata are among the best known insects, our knowledge of detailed species distribution, especially of rare species, was still very limited. The main source of information was Cannings and Stuart (1977), as well as records from a few casual collectors over the intervening years. So, to more accurately rank the dragonflies of the province, a series of surveys were initiated throughout British Columbia.

METHODS

In 1995–98, surveys were conducted in 5 areas of the province: southern Vancouver Island (Hutchings 1997), the Lower Mainland, the Okanagan (Cannings 1998), the northeast corner, and the Columbia/Kootenay area. In the latter region we concentrated on the mountain national parks: Yoho, Revelstoke, Glacier, and Kootenay. The sites visited within these areas were chosen on the basis of historic records, recommendations by locals, and pre-study of topographical maps, and by scouting the region by vehicle.

In order to maximize the chances of recording all species present in an area, surveys were conducted at various times throughout the active odonate flying season, and key sites were visited several times. Some early- or late-flying species have been under-represented in earlier collections because entomologists collect primarily in mid-summer. At each site, voucher specimens were collected of each species observed. Adults were collected using aerial nets. Relative abundance estimates were made for each of the species at a site. When weather conditions would not allow aerial collections, larvae were collected in dip nets. The adult voucher specimens

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

were killed with ethyl acetate, then fixed in acetone for 8–24 hours, dried, and stored in glassine or acetate envelopes until accessioned by the Royal British Columbia Museum. Larvae were placed into 70% ethanol. Specimens were identified using Cannings and Stuart (1977), Westfall and May (1996), Walker (1953, 1958), Walker and Corbet (1975), Cannings (1996), and Paulson (1997), as well as by using comparative material in the Royal British Columbia Museum and the Spencer Entomological Museum at the University of British Columbia.

Habitat descriptions were made for each site where specimens were collected. Wetland type, dominant vegetation, temperature, pH, and amount of disturbance were recorded.

RESULTS

A total of 78 species were collected from more than 550 sites. Eighteen of the 22 species on the preliminary list of rare dragonflies of British Columbia were collected. Changes in rank were made for 10 species based on the results of the surveys. The result is a rare and endangered list of 24 species (Appendix A) and a total provincial list of 85. This includes the addition of 3 new species to the provincial list: *Somatochlora kennedyi*, *S. forcipata*, and *Lestes forcipatus*.

S. kennedyi had been collected at Loon Lake in the Yukon Territory within 3 km of the British Columbia border (Cannings et al. 1991) and from the southern Northwest Territories (Walker and Corbet 1975), so was a target species during the surveys in the northeast. Only 1 individual was recorded: a female, flying near a sedge fen near Andy Bailey Provincial Park just southeast of Fort Nelson.

Edmund Walker collected *S. forcipata* in the 1920s at Boom Creek, Alberta, about 3 km from the British Columbia–Alberta border in Banff National Park. This had been the only record west of Manitoba. By keying in on areas that matched Walker's descriptions of this species' habitat—small, spring-fed boggy streams "following a devious course" in "glades in spruce forests"—3 locations in British Columbia were found: 2 in Yoho National Park, and 1 in Kootenay National Park.

L. forcipatus had been recently found in Washington State, the first record west of Montana (Paulson 1997). In Canada it was not known west of Saskatchewan (Walker 1953). We located it in the Bluewater Creek area, in the Rocky Mountain Trench north of Golden. It is possible that this species may be more widespread, as *L. forcipatus* looks much like the widespread and abundant *L. disjunctus*. The males of the 2 species are virtually indistinguishable.

Two species were encountered frequently enough that they have been removed from the tracking list: *Coenagrion angulatum* (Fig. 1) and *Ischnura erratica*. *C. angulatum* previously had only been known from 1 site in the Dawson Creek area, but in 1997 it was found commonly in ponds, disturbed or otherwise, at a number of sites in northeastern British Columbia. In the rest of its range across the prairies it is a common pond or slough resident. Previous to the 1996 surveys of the Lower Mainland and southern Vancouver Island, *I. erratica* was considered "rare and local throughout its range [in B.C.]" (Cannings and Stuart 1977). The survey turned up this early-flying species in many new localities, including roadside ditches in suburban Vancouver!

Both *Aeshna tuberculifera* and *Somatochlora cingulata* (Fig. 2) remain on the list, but increased information about their range, habitat, and abundance has changed their rarity ranks from S2S3? to S3.

A number of species were either not found or found to be

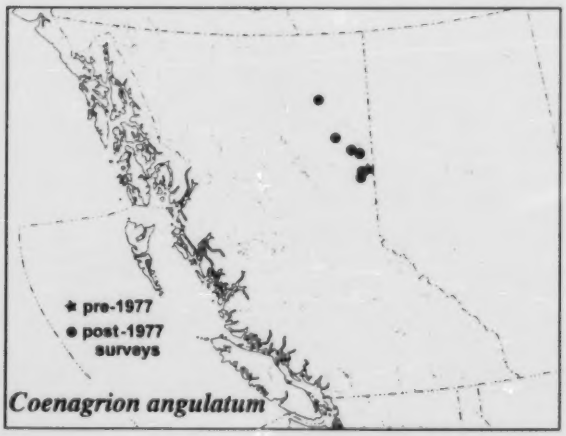

Figure 1. The present knowledge of the distribution of *Coenagrion angulatum* in British Columbia.

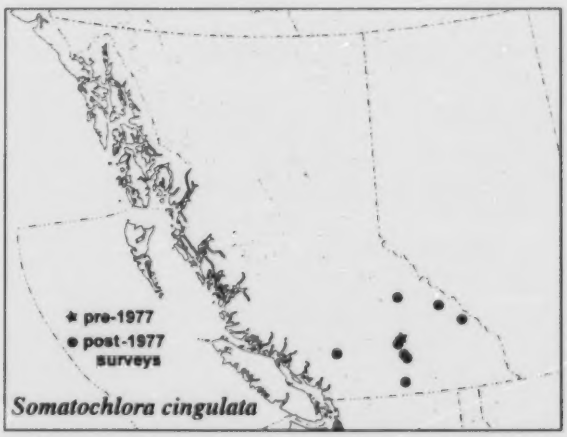

Figure 2. The present knowledge of the distribution of *Somatochlora cingulata* in British Columbia.

not very common or in threatened habitats. The ranks of these species either remained the same or were given a rank denoting that the species was rarer than previously thought. Species that were given a rarer ranking were *Somatochlora hudsonica* (S5 to S4S5), *S. septentrionalis* (S3S4? to S3?), *Erythemis collocata* (S2S3 to S2), and *Leucorrhinia patricia* (S3S4? to S3?). The change in the rank of *E. collocata* changes its provincial list status from Blue to Red.

In addition to increasing our knowledge of the rare species of dragonflies, we learned much more about the distribution, habitat requirements, and flight period of some of the more common species. A few examples of these are given here. *Somatachlora walshii* was only known from half a dozen locations, scattered throughout the southern half of the province. It soon became apparent that it could be found in many spring-fed wetlands with slow-moving water. *Nehalennia irene*, another example of an early-flying species missed by mid-summer entomologists, is now known from the northeast portion of the province, Vancouver Island, and in the Columbia Basin, and for a longer flying season than previously known. When *The Dragonflies of British Columbia* was published in 1977 (Cannings and Stuart 1977), this tiny, iridescent damselfly had only been found in the interior of the province and no further north than Terrace. *Aeshna subarctica* had always been assumed to be more common than the scanty records indicated, and these surveys confirmed this with many records from the Columbia Valley/Kootenays and northeastern portion of the province. Even though it was known from the southern Yukon (Cannings et al. 1991), *Somatochlora minor* had been only found in British Columbia south of 52°N. Several records now exist from as far north as the Fort Nelson area.

DISCUSSION

These surveys were done on a relatively small scale, primarily by 1 or 2 people for a limited amount of time over a single season, but even with this level of effort and with many more areas to survey, places to revisit, and species to look for, they have resulted in a much more defensible Red and Blue list.

Because we were able to visit some key sites repeatedly, we were able to obtain a good estimate of the relative numbers of each species, as well as good species lists for those sites. These initial baseline data can thus be used to monitor populations of dragonflies at these sites. There are also data to monitor specific species over a number of sites.

When the assemblages of observed dragonflies and their habitats are associated, one can begin to predict what species will occur in which habitats within a certain geographical range. This can be applied to the rare as well as the common species. The reverse is also true—certain dragonfly species can be used to predict wetland type. For example, if an *Argia vivida* is found, one can be certain that the waters are spring-fed. Used in conjunction with vegetation, the presence of rare ecosystems can be predicted and defined.

A less tangible but very beneficial result was the fostering of interest amongst amateurs. We led dragonfly field trips for local naturalists and presented slide shows and offered informal information sessions. In addition, the results of these and other surveys will be gathered together to aid in the production of a field guide to the Odonata of the Pacific Northwest. A comprehensive field guide will make it easier and more rewarding for naturalists to identify dragonflies in the field, encouraging them and allowing them to provide reliable information about the group throughout the region. A parallel field where this has proven to be extremely useful has been ornithology; good field guides have enabled amateur birders to contribute useful and accurate data to the overall knowledge of ranges, habitats, and habits of many birds. Because of the large size of British Columbia and the seasonality of the Odonates, the recruitment of enthusiastic, well-informed amateurs is essential to complete the picture of dragonflies in the province.

These results highlight the value of intensive, targeted surveys and illustrate the dynamic nature of the ranked lists. This is, of course, true for not only this group, but for all taxa. They not only fill in key blank spaces in our knowledge, but also focus future studies on species and regions for which there is still information lacking.

ACKNOWLEDGEMENTS

All of these surveys were joint efforts with a variety of funding sources, volunteers, and agencies. Funding and aid in kind has come from the British Columbia Ministry of Environment, Lands and Parks; Habitat Conservation Trust Fund; Parks Canada; the Columbia Basin Trust through The Living Legacy project of the Royal British Columbia Museum (RBCM); and Forest Renewal British Columbia. Principal investigators have been: Richard Cannings, Robert Cannings, S. Cannings, G. Hutchings, R. Kenner, H. Nadel, L. Ramsay, and D. St. John. Additional collectors have included, but are not limited to, G. Archard, Russell Cannings, S. Coates, B. Ferguson, D. Fraser, L. Halverson, N. Newhouse, D. Nicholson, and J. Woods. B.C. Parks and Parks Canada have provided permits for collecting. D. Paulson has helped with advice and identifications, and D. Blades from the RBCM has helped with databases. Many, many others have enthusiastically helped out by swinging nets, pointing out good wetlands, and providing access to areas.

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Appendix A. The rare Odonata of British Columbia.

Scientific name	English name	Global	Provincial	
		Rank	Rank	List
FAMILY LESTIDAE				
Lestes forcipatus	Sweetflag Spreadwing	G5	S1S3	RED
FAMILY COENAGRIONIDAE				
Argia emma	Emma's Dancer	G5	S3S4	BLUE
Argia vivida	Vivid Dancer	G5	S2	RED
Enallagma civile	Familiar Bluet	G5	S1	RED
Enallagma hageni	Hagen's Bluet	G5	S3S4	BLUE
Ischnura damula	Plains Forktail	G5	S1	RED
FAMILY AESHNIDAE				
Aeshna constricta	Lance-tailed Darner	G5	S2S3	BLUE
Aeshna tuberculifera	Black-tipped Darner	G4	S3	BLUE
FAMILY GOMPHIDAE				
Gomphus graslinellus	Pronghorn Clubtail	G5	S2	RED
Octogomphus specularis	Grappletail	G4	S2	RED
Ophiogomphus colubrinus	Boreal Snaketail	G5	S3?	BLUE
Stylurus olivaceus	Olive Clubtail	G4	S2	RED
FAMILY PETALURIDAE				
Tanypteryx hageni	Black Petaltail	G3	S2S3	BLUE
FAMILY MACROMIIDAE				
Macromia magnifica	Western River Cruiser	G4	S3	BLUE
FAMILY CORDULIIDAE				
Epitheca canis	Beaverpond Baskettail	G5	S2S3	BLUE
Somatochlora cingulata	Lake Emerald	G5	S3	BLUE
Somatochlora forcipata	Forcipate Emerald	G5	S1S2	RED
Somatochlora kennedyi	Kennedy's Emerald	G5	S1S2	RED
Somatochlora septentrionalis	Muskeg Emerald	G5	S3?	BLUE
FAMILY LIBELLULIDAE				
Erythemis collocata	Western Pondhawk	G5	S2	RED
Leucorrhinia patricia	Canada Whiteface	G4	S3?	BLUE
Libellula pulchella	Twelve-spotted Skimmer	G5	S3	BLUE
Pachydiplax longipennis	Blue Dasher	G5	S2S3	BLUE
Sympetrum vicinum	Yellow-legged Meadowfly	G5	S3S4	BLUE

Butterflies of Conservation Interest in Alberta, British Columbia, and Yukon

Norbert G. Kondla
British Columbia Ministry of Environment, Lands and Parks
845 Columbia Avenue, Castlegar, BC, V1N 1H3, Canada
Norbert.Kondla@gems3.gov.bc.ca

Crispin S. Guppy
British Columbia Ministry of Environment, Lands and Parks
322 Johnston Avenue, Quesnel, BC, V2J 3M5, Canada

Jon H. Shepard
RR 2, Site 22, Compartment 44
Nelson, BC, V1L 5P5, Canada

ABSTRACT

Alberta, British Columbia and Yukon support 78% of the resident Canadian butterfly fauna. Our knowledge of butterfly taxonomy and distribution in this area is reviewed. Five specific areas and several taxa are identified as priorities for additional work. Habitat is the real issue. Co-operative land use decisions that stress ecosystem management, supplemented by single species management where warranted, will likely be the most effective strategy to retain butterflies at risk.

Key words: Alberta, butterflies, distribution, habitats, insect, taxonomy, Yukon.

Butterflies are the "charismatic megafauna" of the insect world. In the parlance of conservation optics, the phrase "charismatic microfauna" can be used to reflect the growing popularity of butterflies as subjects of recreational fulfilment, scientific enquiry, and conservation efforts. New et al. (1995) refer to butterflies as important flagship taxa for invertebrate conservation. This paper explores the faunal context of the butterflies of Alberta, British Columbia, and Yukon, mentions past work, identifies several areas that are deemed to be of primary conservation interest and discusses management issues and strategies.

Scott (1986) provides context for the butterflies of this area through his range maps for the butterflies of all of North America. Layberry et al. (1998) provide an updated treatment of butterflies of Canada, Bird et al. (1995) provide a detailed treatment of the butterflies of Alberta, Lafontaine and Wood (1997) summarize the butterflies of Yukon, and Shepard and Guppy (in prog.) supply a detailed treatment of the butterflies of British Columbia. Only a few sources of information are specific to butterfly fauna and conservation in the areas of primary conservation interest in British Columbia and Alberta: Smith and Bird (1977), Thormin et al. (1980), Bird (1982), Guppy and Shepard (1994), Guppy et al. (1994), Kondla et al. (1994), Shepard (1995), St. John (1996), and Kondla (1998).

The literature on conservation of butterflies has grown substantially recently. Pyle et al (1981) and Samways (1993) address insect conservation with many butterfly examples, while Pyle (1976), Hammond and McCorkle (1984), New (1991), Pollard and Yates (1993), Gaskin (1995), and New et al. (1995) discuss butterfly conservation specifically.

FAUNAL AND GEOGRAPHIC CONTEXT

Insects constitute between 75 and 85% of the animal kingdom species diversity. Globally, Gaston (1991) reviewed past estimates of species diversity in insects and provided a plausible analysis to arrive at an estimate of 5 million described and undescribed insect species in the world. In North America there may be >150,000 species and in Canada the number of insect species has been estimated at 55,000 (Danks 1978).

As pointed out by Gaston (1991), the Lepidoptera are generally regarded as the best collected and studied of the 4 major insect orders. Six of the 7 families of butterflies in this order occur in Canada: Hesperiidae, Papilionidae, Pieridae, Lycaenidae, Riodinidae, Nymphalidae. Layberry et al. (1998) report that 293 species of butterflies have been found in Canada. Of these, 24 only irregularly or rarely find their way into Canada as migrants or strays and hence are not viewed as part of the resident fauna.

In comparing the butterfly species diversity of Alberta, British Columbia, and Yukon to the national picture, we see

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 498pp.

the following interesting numbers:

- this area (21% of the Canadian land base) supports 78% of the resident Canadian butterfly fauna;
- Alberta and British Columbia comprise only 16% of the Canadian land base but contain 76% of the resident Canadian butterfly fauna;
- 24% of Canadian butterfly species are only found in Alberta, British Columbia, and Yukon;
- 25 species are found only in British Columbia;
- 19 species are found only in British Columbia and Alberta;
- 2 species are found only in British Columbia and Yukon;
- Alberta and Yukon each have 1 unique species.

These numbers will change due to pending and future taxonomic changes but they show the very significant contribution that extreme western Canada makes to species-level diversity. This exceptional level of species richness in butterflies is consistent with the results reported by Pojar (1993) for other groups of plants and animals and is not surprising when considering the geological, elevational, climatic, and ecological diversity of this part of Canada.

TAXONOMY AND DISTRIBUTION

Although imperfect and incomplete information should not be used to justify inaction in butterfly conservation, we should proceed with humility and recognize that taxa believed to be imperilled may in fact not be so. This is an important consideration in western Canada, where our knowledge of fundamentals like taxonomy and distribution is not what it should be to support definitive assessments of conservation status.

Distribution and abundance information is key to assessing which taxa may be at risk. Wrong decisions can lead to costly restrictions on land and resource use as well as directing scarce resources towards taxa that are not at risk while others become more imperilled or even extinct. An example of this situation is the designation of the West Virginia white (*Pieris virginiensis*) as an endangered species in Ontario in the 1970s. Subsequently, many new and stable populations were discovered in the 1980s and the species was wisely removed from the endangered species list in 1990 (Layberry et al. 1998). During the same period, 1 species (frosted elfin, *Callophrys irus*) and 1 subspecies (Karner blue, *Lycaeides melissa samuelis*) became extirpated from Ontario and Canada, through lack of appropriate habitat management.

Both the taxonomy and nomenclature of North American butterflies have been dynamic over the past 20–30 years. There will be additional changes over the next 10–20 years before the dust settles. The situation is not as stable as workers on birds and mammals are accustomed to. Note that this instability is due to rapidly advancing scientific knowledge. Samways (1993) observes that: "Many species are being separated and others synonymized, all within the slippery realms of not knowing exactly when a species is a good species." However, if conservation efforts are postponed until all the taxonomic issues are resolved, it may be too late for some species.

Some people are not fond of the subspecies concept, especially where phenotypic variation is gradual and continuous across the landscape. However, conservation of genetic material below the species level is an important and useful component of insect conservation. Like it or not, conservation decisions are ultimately political decisions. Having a scientific and common name to apply to a recognizable set of populations or metapopulations that are at risk, where scientifically sound, is helpful in seeking the necessary decisions. Good luck to anyone who strives to bring scarce resources to bear on an imperilled but unnamed "bug."

We need to be clear on the source and geographic scale of our existing information so that it is not used to draw erroneous conclusions. Distribution maps for Canada (Layberry et al. 1998) portray some areas of the country with numerous dots, thereby suggesting a high level of sampling intensity in such areas. While this is true relative to other, less intensely sampled areas, it is not true in absolute terms. Mostly these distribution dots (like those in other Canadian butterfly publications) simply mean that a butterfly enthusiast saw or caught at least 1 specimen while exploring a few hectares of land on a sunny afternoon or while making a 20-minute roadside stop at a location that they perceive as likely having some butterflies. Some arithmetic would likely show that about 99% of Canada has never seen a butterfly net nor a person that can accurately identify more than a few common butterfly species. This is especially so in the large area covered in this paper, with few qualified observers being out and about regularly. Seeing one or a few dots on a published distribution map does not mean the taxon is at risk. All this means is that we need to do further work to see if it really is at risk and hence worthy of our efforts to retain it as a component of our biodiversity.

What this means in a practical sense when dealing with small organisms living in a large landscape is that butterflies believed to be rare or even extinct can show up unexpectedly. Some examples are:

- An undescribed subspecies of the large marble (*Euchloe ausonides* ssp.) known in Canada only from the extreme southern Vancouver Island area was believed to be globally extinct. Fortunately, the world does not stop at political borders. A recent discovery of this subspecies in nearby Washington State means that it is not globally extinct and opens the door to possible future reintroduction to Canada.
- Fender's blue (*Icaricia icarioides fenderi*), a resident of native grasslands in the Willamette Valley of western Oregon, was believed to have been extinct since 1937; however, a surviving colony was found in 1989 (Hammond 1995).

- The hobomok skipper (*Poanes hobomok*) was not reported from Alberta until the late 1970s (Bird and Smith 1979), even though it was subsequently found to reside across the road from the campus of the University of Alberta which has employed entomologists since the early 1900s.
- The eastern tailed blue (*Everes comyntas*) was for some years known in British Columbia from only 1 remote locality in the east Kootenays; in the past 2 years we have discovered a sizeable population in the west Kootenays which has a 100-year history of resident butterfly enthusiasts.

The ease with which small and visually ephemeral organisms can be overlooked by even experienced researchers is illustrated by the following example. An often-sampled spot near Atlin, B.C., was explored by parking in a public pull-out and walking up the mountain slope. It was not until one of us parked in the same place and walked down the slope that we discovered a population of a butterfly previously unknown in the northern half of British Columbia.

Only long term and repetitive field exploration will improve our knowledge of what lives where. As pointed out by Gaskin (1995), "rare" can be just a reflection of surveys not done in the right place, not done in the right months, or not done in the right years.

HABITATS AND BUTTERFLIES AT RISK

Despite our limited knowledge, we can confidently identify the following 5 areas in Alberta and British Columbia that warrant recognition as hot spots for butterfly conservation:

1. the extreme south coast of British Columbia, especially southeastern Vancouver Island;
2. the southern Interior of British Columbia, especially the southern Okanagan Valley and the adjacent Similkameen Valley;
3. the extreme southeast corner of British Columbia and immediately adjacent corner of southwest Alberta;
4. the Peace River lowlands, especially the Peace River Valley of Alberta and British Columbia; and
5. the lower Milk River area of extreme southern Alberta.

The south coast of British Columbia is home to about 75% of the human population of the province and all but the steepest habitats are open to the sundry forms of development needed to sustain a burgeoning human population. Those remaining areas not directly impacted by construction or agriculture are then negatively impacted by fragmentation, invasion of introduced weeds, wildfire suppression, and the press of humanity using such "green spaces" for recreation. This area should clearly remain a priority for further research and, more importantly, the development and implementation of conservation actions. Several taxa are under status review by COSEWIC (Committee on the Status of Endangered Wildlife in Canada): large marble (coastal subspecies), Taylor's checkerspot (*Euphydryas editha taylori*), island checkerspot (*Euphydryas chalcedona perdiccas*), greenish blue (*Plebejus saepiolus insulanus*), and dun skipper (*Euphyes vestris*).

The south Okanagan Valley and lower Similkameen Valley are also subject to the increasing press of human activity and rank as a close second in terms of urgency for further attention to the locally distributed butterflies of this area. Wholesale habitat conversion and the effects of aggressive weeds are the primary threats to several butterflies. Behr's hairstreak (*Satyrium behri*) is under review by COSEWIC and a few others, such as Mormon metalmark (*Apodemia mormo*) and sooty hairstreak (*Satyrium fuliginosum*), should be formally assessed as well. The monarch (*Danaus plexippus*), which breeds in a wider area of the southern Interior, is already listed as "vulnerable" by COSEWIC.

The situation in extreme southeastern British Columbia and southwestern Alberta is not as bleak as the 2 previous areas; although the only known British Columbia population of the grey copper (*Lycaena dione*) is at high risk from urban development at Cranbrook. Long-term threats to the survival of localized taxa emanate from fire suppression and excessive levels of livestock grazing. Well-planned logging, grazing, and revegetation of disturbed areas can even be helpful in retaining populations of some butterflies such as Gillette's checkerspot (*Euphydryas gillettii*) and eastern tailed blue.

The lower Milk River area of southern Alberta seems to be in generally good shape insofar as there is still substantial natural habitat for those butterflies for which we have relatively few records. Long-term threats include flooding of riparian areas, excessive livestock grazing, and inappropriate location of oil and gas industry infrastructure. COSEWIC is reviewing the status of Weidemeyer's admiral (*Limenitis weidemeyeri*).

The Peace River area, especially the Peace River Valley of Alberta and British Columbia, is a real sleeper in terms of significance for butterfly conservation. It does not have the intensity of human activity present in other areas and most of the land use impacts are a "done deed." However, it does contain a number of highly localized butterfly phenotypes, most of which remain to be critically examined taxonomically. Unlike most of the other taxa that are of conservation interest, these phenotypes are not found in other political jurisdictions and hence their fate rests in our hands. Fire suppression has changed the natural disturbance ecology of the remaining grasslands and shrublands of the residual natural habitats. Continued "unnatural" succession will not bode well for the long-term survival of taxa that require grassland (e.g., Alberta Arctic, *Oeneis alberta*) or that may have evolved to survive in a natural disturbance regime that produces juxtaposed grassland, shrubland, and forest patches (e.g., northern checkerspot, *Chlosyne palla* ssp.). Establishment of protected areas such as parks and ecological reserves will not assure the sur-

vival of these butterflies in the future.

We include Yukon in this brief review primarily to provide a written reference to a good news scenario. Persons unfamiliar with butterflies should not consider any Yukon butterflies as being at risk based solely on a few known records. Much additional fieldwork will be needed to clearly define the distribution and habitats of Yukon butterflies. The scant human population in relation to the size of the land base and the nature of contemporary land use means that there is no cause for concern in the foreseeable future. The only butterfly conservation issue we are aware of in Yukon is periodic heavy grazing of the unique Duke River grasslands, and resulting possible reduction in populations of grass-feeding species such as the draco skipper (*Polites draco*).

MANAGEMENT IMPLICATIONS

We should not lean heavily on legislated prohibition of direct human predation (collecting) as an effective means of ensuring butterfly conservation. We agree with the observations made by Gaskin (1995) on this matter:

- It gives the illusion of decisive remedial action while the real culprits of habitat destruction and degradation continue unabated.
- Efforts of amateur and professional entomologists to undertake basic research are discouraged by intricate regulatory requirements.
- Poorly framed regulations, especially those with unnecessarily punitive prohibitions, cause more problems than they solve, especially without habitat protection.
- Overzealous application of poorly framed laws alienates landowners, discourages much needed volunteer fieldwork, and creates an atmosphere of distrust that prevents the kind of cooperative relationships needed for effective conservation.
- Of course having legislation that deals with outright poaching of endangered species is valuable.

Habitat! Habitat! Habitat! This timeless refrain rings as true with butterflies as it does with other animals and plants, both in identifying the cause and the cure for butterflies at risk. Remorseless habitat destruction and degradation by *Homo sapiens* has unquestionably reduced the historical range and population levels of some butterflies. Of course, human activity has also expanded the range and increased the populations of other butterflies—but the overall trend is clearly downward.

Given the reality of human subsistence needs and recreational desires, the only issue that will have a measurable impact on butterfly conservation (at least in our corner of the world) is land use. We will not be able to retain butterflies at risk without making land use decisions that provide for their continued existence.

Habitat destruction and degradation are not limited to immediate and obvious wholesale ecosystem changes such as cultivating a piece of natural grassland, new housing developments, or clearcutting a patch of old seral forest. Replacement of native vegetation by aggressive weeds is insidious and equally as serious a threat to natural ecosystems. Even the simple and popular act of planting trees can be a disaster. Witness for example the major contribution to extirpation of the Karner blue in Canada by "reforesting" Pinery Provincial Park in Ontario. By creating a closed canopy forest in a landscape that was never naturally forested, at least not in historical times, the well-meaning architects of this new forest unintentionally destroyed not only the Karner blue and other endangered organisms but an entire natural ecosystem (Cundiff 1995).

An example of a decision that may have unintentionally benefited butterflies is the story of a dense population of Mormon metalmarks at Keremeos, B.C. A historical decision to construct a railroad at the base of a slope and subsequent mechanical disturbance of the slope may have resulted in a dramatic increase in abundance of the butterflies' larval food plant (snow buckwheat, *Eriogonum niveum*). Highway construction has created more cutbank habitat for additional subpopulations. A viable metapopulation of this butterfly could likely be ensured in the lower Similkameen Valley by artificially increasing food plant density on suitable southerly facing slopes, ensuring the presence of nectar sources and keeping livestock grazing intensity to a reasonable level.

The primary strategy for conservation of butterflies at risk is to pursue the decision-making fundamentals of information, choice, action, and evaluation of actions. Some people call this adaptive management but the good sense approach of making decisions and evaluating them has been in existence for a long time. This simple strategy has the advantage of being independent of geographic scale, time, and the nature of the decision being made.

Sufficient information exists to narrow the geographic scope of further information collection to the 5 areas mentioned in the previous section. In these areas we need to determine the exact locations and habitats of the taxa that appear to be at risk, likely threats to their continued existence, population trends, and habitat trends. At the same time we should continue to more fully investigate the status of many other taxa outside of these areas which are known from relatively few sites (Guppy et al. 1994, Kondla 1998) but for which no evidence supports any "at risk" designation.

Most taxa that appear to be at risk in our area are in fact "corner weeds" that are widely distributed in North America. Risk management suggests that maintaining populations of endemic organisms is more important than maintaining a peripheral population of an organism that is "rare" only through the view of political boundaries. We do not suggest that we do nothing about the peripheral gene pools but rari-

ty created by political boundaries should certainly influence our decisions on priorities and allocation of resources.

No single person or institution in our society has the power and resources needed to act unilaterally in retaining our butterflies at risk. Good science and good legislation will not be enough. Good information sitting in a computer or office will not be enough. Establishing various kinds of protected areas will not be enough. What will be enough will be to have situational and strategic partnerships among the relevant decision-makers. These partnerships will need to operate primarily by ecosystem management and selectively exercise single species management where a combination of science and social choice point to such an approach. The partnerships will need to encompass the gamut of individual private land owners, corporate land owners, First Nations, municipal government, federal government and not-for-profit conservation organizations.

As we go about doing good things for species at risk, we should maintain humility and a larger perspective. Some points to ponder from a thought-provoking article by McFadden and Parker (1994) are:

- Of all the species that have ever existed, 99% are extinct.
- Extinction is a normal part of the evolutionary process.
- We humans are just another component of the species mix on planet Earth.
- We have become a major system perturbation that could eventually have impacts as great as all the natural events that have ever occurred.
- Extinction will continue, and so will speciation and biological diversity.

How much of an increased extinction rate due to human activity will there be? There will be some. The amount of extinction that we can influence will be determined through value-driven social choice and the financial capacity of our collective institutions.

ACKNOWLEDGEMENTS

We thank the following people for their help in preparing this paper: G. Hilchie, D. Lafontaine, E. Pike, C. Schmidt, C. Wallis. As well, we acknowledge the support of the British Columbia Ministry of Environment, Lands and Parks in the preparation and presentation of this paper. Views expressed herein are those of the authors only.

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The Potentially Rare and Endangered Terrestrial Arthropods in British Columbia: Revisiting British Columbia's Biodiversity

Leland M. Humble

Natural Resources Canada, Canadian Forest Service
506 West Burnside Road, Victoria, BC, V8Z 1M5, Canada
lhumble@pfc.forestry.ca

Neville N. Winchester

Biology Department, University of Victoria
P.O. Box 3020, Victoria, BC, V8W 3N5, Canada

Richard A. Ring

Biology Department, University of Victoria
P.O. Box 3020, Victoria, BC, V8W 3N5, Canada

ABSTRACT

This paper provides an update to portions of the annotated systematic list of the endemic and potentially rare and endangered terrestrial arthropods in British Columbia published by Scudder (1994). Detailed arthropod diversity inventories in forest sites in coastal British Columbia have provided significant new information on the presence and distribution of species in these habitats. Since 1991, more than 3.4 million specimens have been collected; they form 1 of the largest databases in North America to answer biodiversity-driven questions concerning arthropods, endemism, disjunct distributions, dispersal capabilities, and the impact of exotic species. We present evidence to support the assertion that these forests are global and continental foci of diversity for several taxa. The summarizing of these basic patterns of species richness and distribution for arthropods forms the framework that is at the heart of the biodiversity challenge. This information must be considered if the issues that surround the maintenance of biological diversity in British Columbia is to be integrated and interpreted at all taxonomic levels.

Key words: Araneae, arthropod, Coleoptera, nonindigenous, old-growth forests, Oribatida.

Identification of the components of biological diversity important for conservation cannot be accomplished without an assessment of endemic, rare, and threatened species. Scudder (1994) compiled the first major annotated, systematic list of potentially rare and endangered invertebrates in British Columbia based on existing published information and consultation with experts on each group. His work identified a total of 168 species endemic to the province and 448 rare species that, within Canada, occur only in British Columbia. An additional 203 more widely distributed species were listed on the basis of their limited occurrence or their restriction to threatened ecosystems. He noted that ongoing inventory work and research on collections, which were for the most part not consulted in the original study, would result in future modifications of the list.

Our current understanding of the conservation status of the arthropod fauna of British Columbia is derived from existing publications, current taxonomic knowledge, and frequency of occurrence of individual species in collections. Catalogues that record the regional occurrence in Canada or North America (by province and state) are available for major arthropod taxa such as the beetles (Bousquet 1991), bees, wasps, sawflies, and parasitic wasps (Krombein et al. 1979), flies (Stone et al. 1965), true bugs (Henry and Froeschner 1988), and oribatid mites (Marshall et al. 1987, Behan-Pelletier 1993). Additional provincial records can be found in regional and provincial checklists, as well as lists of new regional or provincial records summarized by Scudder (1994). However, no comprehensive listing of the distribution and abundance of most groups of arthropods is available on a provincial scale. Taxonomic or ecological research studies on specific groups (e.g., dragonflies and damselflies [Cannings and Stuart 1977]; butterflies [Guppy et al. 1994, Layberry et al. 1998]) or in specifically threatened ecosystems or habitats (Scudder 1992, 1993) provide detailed information on the distributions or the conservation status of these groups.

We present data from detailed arthropod diversity inventories in forest sites on Vancouver Island and the lower mainland, which provide significant new information on the presence, distribution, and abundance of species in these habitats. Since 1991, more than 3.4 million specimens have been collected. Identifications of many arthropod taxa

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

collected in these studies are continuing. However, the identifications completed to date form 1 of the largest data sets in North America to answer biodiversity-driven questions concerning arthropods, endemism, disjunct distributions, dispersal capabilities, and the impact of exotic species. The information we present here is derived from specific taxa: the beetle mites (Oribatida); 2 families of beetles (Coleoptera)—the rove beetles (Staphylinidae and related families) and the bark and ambrosia beetles (Scolytidae); and the spiders (Araneae) for which we have a high degree of species resolution.

METHODS

These studies were carried out in ancient and regenerating coniferous forests in the Upper Carmanah Valley (CV; 48°42'N, 124°38'W) on the west coast of Vancouver Island; in coniferous forests at Rocky Point (RP; 48°20'N, 123°35'W) near Victoria; in montane old-growth forests at Mount Cain (MC; 50°14'N, 126°20'W), north of Campbell River; and in montane old-growth and selectively harvested forests at the Montane Alternative Silviculture Systems (MASS) research site (49°51'N, 125°26'W), south of Campbell River on Vancouver Island. These research studies were undertaken to determine the arthropod diversity of ancient and regenerating forests. Collecting techniques used to generate the datasets at CV, RP, and MC included passive trapping (Malaise and pan traps), as well as direct sampling of branches and suspended moss mats (CV only). The MASS datasets were generated from specimens collected using passive trap systems (Malaise and window traps) and active trap systems (light and baited Lindgren funnel traps), as well as from direct sampling of branches and arboreal lichens.

RESULTS AND DISCUSSION

Coleoptera

More than 54,000 beetle specimens representing >400 species have been identified from trap collections at MASS. A total of 3,335 adult beetles representing 123 identified species were recovered in the CV studies.

Staphylinidae

The rove beetles were the numerically dominant family of beetles recovered from CV (59% of both individuals and species) and from MASS (75% of the individuals, 35% of the species). Sixty percent of the species of Coleoptera and 61% of the Staphylinidae collected at MASS were represented by 5 or fewer individuals in the collections.

Scudder (1994) recorded 37 species of potentially rare and threatened species of Staphylinidae (as defined by Lawrence and Newton 1995), 6 of which were considered endemic. Collections from CV and MASS have resulted in range extensions for 5 of those previously listed species (Table 1). With the exception of *Micropeplus brunneus*, <5 individuals of each of the species designated by Scudder (1994) were collected at the respective research sites. The large numbers of *M. brunneus* collected from the MASS site serve to illustrate the inadequacy of our current knowledge of the arthropod fauna of the province. This species was denoted as rare and endangered in British Columbia, as it was known from only 3 localities and had not been collected in more than 40 years (Scudder 1994). Our data would indicate that this species is more common than previously thought in the montane old-growth forests of Vancouver Island.

An additional 27 described species and an equal number of undescribed species of Staphylinidae were recorded from British Columbia for the first time (Bousquet 1991) at 1 or both of the study locations. Most of these taxa were represented by 5 or fewer individuals (Table 1). While some of these may represent rare and threatened endemics to British Columbia or Canada, their status cannot be determined until taxonomic revisions for each taxon are available.

Scudder (1994) included *Xylodromus depressus* as a potentially rare and endangered species because of its limited occurrence in British Columbia, although it does occur elsewhere in Canada. It is our contention that it should be removed from the listing, as Campbell and Davies (1991) note that it is not native to North America. Another 5 nonindigenous species of rove beetles were documented to be present at 1 or the other study location (Table 1). Most of these exotics are uncommon in the forest habitats sampled, being represented by <2 individuals in the collections. However, the numbers of 1 of the exotics captured at MASS, *Trichophya pilicornis*, were surprising, as <24% of the native Staphylinidae were as abundant. The overall abundance of *T. pilicornis*, and its presence at 4 of the 5 locations sampled, suggest that it is strongly invasive in these forest habitats. Feeding habits are known for only 4 of the introduced taxa; all are predators on other arthropods.

Scolytidae

The North American scolytid fauna is well known taxonomically. McNamara (1991) records 204 species of bark and ambrosia beetles from Canada and Alaska, with 65% of the species present in the Canadian fauna occurring in British Columbia (134 species). Of these, Scudder (1994) lists a single species, *Hylocurus hirtellus*, as potentially rare and endangered in British Columbia, based on a single collection record from Victoria (Bright and Skidmore 1997). This collection represented the only record for this species in Canada, although it ranges as far south as Baja California (Scudder 1994). During recent research on nonindigenous introductions in the greater Vancouver area, *H. hirtellus* was collected in traps or reared from 4 locations separated by >35 km (Ladner to Langley). This native ambrosia beetle

Table 1. Potentially rare and threatened and/or introduced Staphylinidae (as defined by Lawrence and Newton 1995) from 2 arthropod diversity studies on Vancouver Island. Where listed, the designation of status given for each taxon in British Columbia is that of Scudder (1994): • indicates that the species occurs only in British Columbia in Canada, but is recorded from elsewhere in North America; and •• indicates that the species is endemic to British Columbia. Numbers of adults of each species are given for the Carmanah Valley (CV) and Montane Alternative Silviculture Systems (MASS) research sites. Total number of identified Staphylinidae recovered from the 2 sites are 1,967 and 54,490 adults respectively. First records for British Columbia (BC) and Canada are denoted by "•" in the appropriate column. Introduced species (Campbell and Davies 1991) are designated "I" under origin.

		Research site		First record		
Status	Taxon	CV	MASS	BC	Canada	Origin
•	*Micropeplus brunneus* Macklin		69			
•	*Nanobius serricollus* (LeConte)		1			
•	*Orus punctatus* Herman	1				
•	*Pseudohaida rothi* Hatch	5				
•	*Stenus ellipticus* Casey	1				
•	*Tanyrhinus singularis* Mannerheim	1				
••	*Trigonodemus fasciatus* Leech	1				
•	*Trigonurus sharpi* Blackwelder	2	4			
	Stictalia (=*Bolitochara*) *californica* Csy		1	•		
	Autalia riculas Gravenhorst		1	•	•	
	Orus sinuatus Herman	2	1	•	•	
	Phloeopora testacea Mannerheim		4	•	•	
	Quedius (*Microsaurus*) *simplex* Hatch	4		•	•	
	Cordalia obscura (Gravenhorst)		2	•		I
	Gabrius nigritulus (Gravenhorst)		1			I
	Gabrius subnigritulus (Reitter)		1			I
	Trichophya pilicornis (Gyllenhal)		17			I
	Xylodromus concinnus (Marsham)		2			I
•	*Xylodromus depressus* (Gravenhorst)	1				I

breeds in the dead wood associated with scars on living host plants and has been reared from 2 hosts, *Salix scouleriana* and *Populus* sp., which are widely distributed across the province. Further inventories of our native bark and ambrosia beetles will be necessary to determine the abundance, and hence the status, of this potentially rare and endangered species in British Columbia.

While only 1 native scolytid is listed as potentially rare and threatened in British Columbia, recent studies have shown that our fauna is being altered. Surveys of the Scolytidae in the greater Vancouver region demonstrate that exotic bark and ambrosia beetles are accumulating in the provincial fauna. Since 1995, 5 recently introduced species of Scolytidae have been discovered in the urban and managed forests of the lower mainland (Humble unpubl. data). These additions have doubled the number of introduced Scolytidae known to occur in British Columbia. Today, >7% of the species of Scolytidae (10 species) present in British Columbia are introductions (McNamara 1991, Wood and Bright 1992).

Surprisingly, introduced scolytids were found at both the MASS and CV research sites. A single adult of *Xyleborus dispar* was recovered from among the 3,341 adult Scolytidae (19 species) recovered at the former study site. Although a limited number of Scolytidae (26 individuals of 6 species) were recovered from the latter research site, 1 individual of a second non-indigenous species, *Xyleborinus saxeseni*, was present.

Table 2. Comparison of the number of families, genera, species, and individuals of arachnids at 4 biodiversity study sites on Vancouver Island, B.C. (MASS = Montane Alternative Silviculture Systems research site.)

Site	No. of families	No. of genera	No. of species	No. of individuals
Rocky Point	15	50	76	3,093
Carmanah Valley	16	53	74	2,179
Mount Cain	16	62	78	1,699
MASS	16	47	68	3,774
TOTAL (all sites)	18	74	167+	10,745

Araneidae

The spider fauna has been well documented, with almost 11,000 individuals representing >167 species having been collected across the 4 research sites (Table 2). Eleven of these species were identified as rare and occurring only in British Columbia, within Canada (Scudder 1994), and an additional 3 species were identified to be rare in British

Table 3. Potentially rare and threatened and/or introduced spiders (Araneae) present (•) at 4 arthropod diversity studies on Vancouver Island. Where listed, the designation of status given for each taxon in British Columbia is that of Scudder (1994): a blank indicates that the species is rare in British Columbia but occurs elsewhere in Canada; • indicates that the species occurs only in British Columbia within Canada, but is recorded from elsewhere in North America; •• indicates that the species is endemic to British Columbia; and n.a. indicates that status is not applicable. Introduced species (D. Buckle, pers. comm.) are designated "I" under origin.

		Research site				
Status	Taxon	RP	CV	MC	MASS	Origin
	AGELENIDAE					
	Agelenopsis actuosa (Gertsch & Ivie)	•				
	Cybaeus signifer Simon	•	•	•		
*	*Ethobuella tuonops* Chamberlin & Ivie	•	•	•	•	
n.a.	*Tegenaria agrestis* (Walckenaer)	•				I
n.a.	*Tegenaria gigantea* Chamberlin & Ivie	•				I
	ANTRODIAETIDAE					
*	*Antrodiaetus pacificus* (Simon)	•	•	•	•	
	ANYPHAENIDAE					
	Anyphaena aperta (Banks)	•	•	•		
	ARANEIDAE					
n.a.	*Araneus diadematus* Clerck	•				I
	CLUBIONIDAE					
*	*Castianeira walsinghami* (O.P.-Cambridge)	•				
*	*Trachelas californicus* Banks	•				
	GNAPHOSIDAE					
*	*Sergiolus columbianus* (Emerton)	•				
	LINYPHIIDAE					
	Bathyphantes diasosnemis Fage			•	•	
*	*Bathyphantes orica* Ivie			•		
n.a.	*Lepthyphantes tenuis* (Blackwall)	•				I
*	*Linyphantes nehalem* Chamberlin & Ivie	•				
*	*Linyphantes orcinus* Chamberlin & Ivie	•	•	•		
*	*Linyphantes pualla* Chamberlin & Ivie		•	•		
	ERIGONIDAE					
*	*Walckenaeria columbia* Millidge		•		•	
	MECICOBOTHRIDAE					
*	*Hexura picea* Simon		•			
	PHILODROMIDAE					
n.a.	*Philodromus dispar* Walckenaer	•	•			I
	THERIDIIDAE					
n.a.	*Enoplognatha latimana* Hippa & Oksala	•				I
n.a.	*Enoplognatha ovata* (Clerck)	•				I
n.a.	*Enoplognatha thoracica* (Hahn)	•				I
n.a.	*Theridion bimaculatum* (Linnaeus)	•			•	I
n.a.	*Theridion tinctum* (Walckenaer)	•				I
n.a.	*Theridion varians* Hahn	•				I

Columbia but occurring elsewhere in Canada (Table 3). Two species included in the former category, (*Ethobuella tuonops* and *Antrodiaetus pacificus*), were present at each of the study sites, and a third species, *Linyphantes orcinus*, was present at 3 of the 4 sites. An additional 2 species, (*Walckenaeria columbia* and *Linyphantes pualla*), listed as occurring only in British Columbia in Canada, and 3 species, (*Anyphaena aperta*, *Cybaeus signifer*, and *Linyphantes orcinus*), identified as potentially rare in British Columbia but occurring elsewhere in Canada, were recovered from 2 or 3 widely separated localities. These collections demonstrate that the aforementioned 8 species are more widespread than

previous collections have indicated (West et al. 1984, 1988; Scudder 1994) and may not be rare. Five of the species previously designated as potentially rare in British Columbia were present only at the RP study area, and 2 species, (*Bathyphantes orica* and *Hexura picea*), were present at MC and CV, respectively. Indeed, for *B. orica* and *Trachelas californicus*, these collections represent only the second collections of both species in the province (Scudder 1994). As the distributions of the latter 6 species are restricted in British Columbia, their status should remain as potentially rare pending further research.

Eleven of the species of Araneae collected from the 4 research sites are introductions to the fauna of the province (Table 3). All of the introduced species were present at the RP study site, and 9 were restricted to that location. A single introduced species was recovered at each of the CV and MASS study sites. While the nonindigenous spider taxa are most numerous (14% of the species recovered) at RP, the collection of *Theridion bimaculatum* and *Philodromus dispar* in more remote forest habitats indicates the potential for dispersal of these introduced species.

Oribatida

The oribatid mite fauna of the CV was intensively studied, especially the fauna associated with moss mats in the canopy of Sitka spruce (*Picea sitchensis*) (Behan-Pelletier and Winchester 1998). The arboreal oribatid mite fauna associated with the foliage and canopy lichens of Pacific silver fir (*Abies amabilis*) and western hemlock (*Tsuga heterophylla*) from MASS are currently being examined. More than 29,000 individuals of 24 species of Oribatida (14 families) have been identified from MASS, and 3,868 individuals of 52 species (27 families) from CV. Less than 25% of the 68 species recovered across both sites can be assigned to existing species. Only 2 of the 15 taxa that can be assigned to existing species have previously been recorded from the province (Table 4). An additional 17 genera, which include at least 22 morphospecies, are recorded for the first time for the province, with 3 (*Dendrozetes*, *Paraleius*, and *Crassoribatula*), also representing the first records of occurrence for Canada (Marshall et al. 1987, Behan-Pelletier 1993).

None of the Oribatida listed as potentially rare and endangered (Scudder 1994) were recovered at the study sites. The mite fauna of the province has been incompletely studied, especially the fauna associated with the forest canopy. Application of Scudder's (1994) criteria to assign a status to the described species recovered in these studies would result in an additional 7 species being added to the category of species rare in British Columbia but occurring elsewhere in Canada, and 6 species being considered rare and occurring only in British Columbia within Canada, as all of the taxa are recorded from, at most, 2 locations in the province. Such designations would be premature, as surveys of the canopy Oribatida have been completed at only the 2 study sites discussed.

The knowledge of the arthropod diversity for most taxa and habitats in British Columbia is both incomplete and inadequate for determining the potential status of rare and threatened species. We present new information for selected arthropod groups that illustrates the need for a comprehensive examination of the arthropod taxa of the province. Species included in taxa that are well known taxonomically (e.g., Scolytidae, Staphylinidae, and Araneae) and listed by Scudder (1994) as potentially rare and threatened are now being described as more widely distributed than had previously been known. Other taxa (e.g., *Micropeplus brunneus*) are now shown to be relatively abundant at selected research sites and thus may not merit designation as potentially rare or threatened. For taxa such as the oribatid mites, determination of their status is impossible, as the mite assemblages of terrestrial habitats have not been well studied (Behan-Pelletier 1993, Scudder 1994). Additionally, the taxonomic expertise available for species descriptions and determinations is inadequate to deal with the diversity of the provincial arthropod fauna.

Quantitative data from these collections provide baseline information for comparisons to determine the impacts of various factors (e.g., nonindigenous introductions, habitat loss) that can threaten biological diversity in forest ecosystems. Nonindigenous spiders comprise a significant proportion of the fauna at Rocky Point, the study site nearest to long-term human settlement on southern Vancouver Island, but at present are minor elements of the fauna at the more remote forest sites. Nonindigenous species of Scolytidae and Staphylinidae are also present at the 2 remote forest habitats for which we have data. After habitat loss, nonindigenous introductions are the next most significant cause of loss of biodiversity. Introduced Scolytidae have been demonstrated to predominate in urban forest habitats in greater Vancouver, and some species have recently been discovered invading the adjacent native forests (Humble unpubl. data). Future quantitative collections at our study sites will allow us to determine if these introduced taxa are becoming more prevalent and may allow determination of their impacts on endemic species assemblages.

Knowledge of the abundance of individual species rather than presence/absence data allows refinement of concepts such as rarity and endemism. Although 3 undescribed species of *Mycobates* are present at the MASS study site and have not been recorded elsewhere in the province, they all cannot be considered rare. Adults of *Mycobates* n. sp. 1 comprise 33% of the adult oribatid mite fauna collected from the canopy foliar samples and will likely be found to be common in other montane forest habitats. The other 2 species of *Mycobates* are rare in canopy foliar samples, but are somewhat better represented in canopy lichen samples. Like the

Table 4. Canopy Oribatida (beetle mites) from branches, lichens, and mosses at Carmanah Valley (CV) and Montane Alternative Silviculture Systems (MASS) research sites, representing first records for British Columbia (BC) and/or Canada. Under first records, • designates a first record of occurrence for a species, and •• designates a first record of occurrence for the genus in the specified geographic area.

			First record	
Taxon	MASS	CV	Canada	BC
CAMISIIDAE				
Camisia horrida (Hermann)	•			•
Camisia segnis (Hermann)	•			•
HERMANNIIDAE				
Hermannia gibba (C.L. Koch)		•		•
LIODIDAE				
Platyliodes macroprionus Woolley & Higgins		•		•
CEPHEIDAE				
Cepheus corae Jacot	•			•
Eupterotegaeus rhamphosus Higgins & Woolley		•		•
EREMAEIDAE				
Eueremaeus acostulatus Behan-Pelletier	•	•		•
Eueremaeus aysineep Behan-Pelletier		•		•
Eueremaeus marshalli Behan-Pelletier		•		•
Eueremaeus chiatous (Higgins)		•		•
LIACARIDAE				
Dorycranosus spp.	•	•		••
PELOPPIIDAE				
Ceratoppia sp.1-3		•		••
Dendrozetes n. sp.	•	•	••	••
Parapyroppia sp.		•		••
TECTOCEPHEIDAE				
Tectocepheus velatus (Michael)		•		
OPPIIDAE				
Oppia sp. 1		•		••
AUTOGNETIDAE				
Autogneta longilamellata (Michael)		•		••
THYRISOMIDAE				
Banksinoma sp.		•		••
CYMBAEREMAEIDAE				
Ametroproctus reticulatus Aoki & Fujikawa	•			
Scapheremaeus palustris Sellnick	•	•		•
ORIBATULIDAE				
Eporibatula sp.		•		••
Phauloppia sp.	•	•		••
Crassoribatula sp.	•		••	••
ORIPODIDAE				
Parapirnodus sp.	•	•		••
n. gen. nr. *Parapirnodus*		•		••
SCHELORIBATIDAE				
Paraleius sp.		•	••	••

Table 4. Continued.

Taxon	MASS	CV	First record: Canada	First record: BC
CERATOZETIDAE				
Neogymnobates n. sp.	•			••
Sphaerozetes sp.		•		••
Jugatala tuberosa Ewing	•	•		•
Mycobates punctatus Hammer		•		•
Mycobates n. spp.1-3	•			••
ACHIPTERIIDAE				
Achipteria sp. 1		•		••
Anachipteria sp.		•		••
GALUMNIDAE				
Pilogalumna sp.		•		••

oribatid fauna associated with moss mats in the canopy of Sitka spruce in the Carmanah Valley, species such as *Mycobates* n. sp. 2 and 3 may be threatened should their habitat (canopy lichens) be lost.

It is important that arthropod diversity be studied at the species level; examination of arthropod diversity at the level of the family or other higher taxa provides no insights into rarity or endemism. Such research is dependent on a strong systematic infrastructure, which at the present time is totally inadequate in Canada (Ball and Danks 1993, Behan-Pelletier 1993, Danks and Ball 1993). Increased support for systematics research, collections, and training is urgently needed to provide the necessary tools and expertise for the study of arthropod diversity.

The study of all aspects of diversity including rarity and endemism is dependent on the availability and long-term curation of reference collections. They provide the "raw materials" for systematics and taxonomic advancement, which develop basic tools such as keys and descriptions needed to distinguish the components of biological diversity (Danks and Ball 1993). Many of the arthropods collected in the studies outlined above and other studies throughout the province are undescribed and are currently identified as "morphospecies." It is important that vouchers of these, as well as fully determined species, are deposited in recognized and curated collections at the completion of arthropod diversity studies. The existence and availability of such voucher specimens allow comparisons to be made at the landscape level, even in the absence of specific identifications, and will also allow further refinements of the concepts of rare and threatened arthropods in the province.

ACKNOWLEDGEMENTS

Fieldwork in the Carmanah Valley was supported by operating grants to N. Winchester and R. Ring from the British Columbia Ministry of Forests, Research Branch, and to N. Winchester and S. Cross from Forest Renewal British Columbia (FRBC). Fieldwork at Mount Cain was conducted by N. Winchester and was supported by operating grants from FRBC to the B.C. Ministry of Forests. Research at the Montane Alternative Silvicultural Systems site was supported by Macmillan Bloedel Ltd. and an operating grant from FRBC to L. Humble. The Canadian Forest Service supported laboratory work on the MASS collections. We wish to acknowledge the essential taxonomic contributions of D. Buckle, J. Klimaszewski, J. Campbell, and V. Behan-Pelletier.

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Two Web-Accessible Fungus Databases and Their Relationship to a Preliminary List of Rare Macrofungi for British Columbia

Brenda E. Callan
Canadian Forest Service
Natural Resources Canada, Pacific Forestry Centre
506 West Burnside Road, Victoria, BC, V8Z 1M5, Canada
bcallan@pfc.cfs.nrcan.gc.ca

ABSTRACT

Two databases documenting in detail the vast fungal biodiversity in British Columbia are now accessible on the Internet at http://www.pfc.cfs.nrcan.gc.ca/biodiversity/herbarium. The British Columbia Host-Fungus Index Database was compiled from 60 years of Canadian Forest Service disease identification data, and published literature records. It was designed to reliably document fungi occurring on native plants and other forest substrates in British Columbia, with the exception of agricultural crops and introduced ornamental plants. The Host-Fungus Index holds records of 3,481 fungus species on 1,710 different plant hosts and other substrates. More than 4,690 of these records are verified by herbarium specimens curated at the Pacific Forestry Centre. Records of >20,000 herbarium specimens are housed in a separate Herbarium Database accessible at the same Web site. The Herbarium Database records can be sorted and mapped by region, date, species, collector, and various other categories. The 2 databases were used to augment and verify a very preliminary list of potentially rare macrofungi (large fleshy or woody fungi) in British Columbia. Mycologists are invited to submit additional records in order to either confirm or dispel the beliefs that these fungi are truly rare and not just undercollected. Logistical challenges surround the creation of lists of rare fungi. Field identification may not be feasible. Specimens often have to be cultured, or verified by distant experts. Fleshy fungi soon lose colour, shape, and other features, so detailed field notes must be made from fresh specimens. Contributions from amateur mycological societies enrich our databases, provided that new records are properly documented.

Key words: database, forest pathology, fungus, macrofungi.

HERBARIUM VOUCHER SPECIMEN DATABASE

The Forest Pathology Herbarium at the Canadian Forest Service's Pacific Forestry Centre (PFC) houses an internationally recognized collection of >35,300 preserved and catalogued forest fungi and disease specimens, representing >3,400 different fungal species. It is known internationally by the acronym DAVFP, and is listed in the Index Herbariorum (Holmgren et al. 1990), a catalogue of the world's herbaria. The acronym is derived from "Department of Agriculture, Victoria, Forest Pathology," and is based upon the herbarium's affiliation at the time of its establishment in 1940.

One of the primary uses of DAVFP is for depositing voucher collections of new fungus distribution records. The voucher collections enable current and future researchers to examine biological specimens in order to confirm published research, or quarantine records. Scientists frequently also borrow collections in order to reexamine them using new taxonomic information and research techniques. More than 140 type specimens are also deposited in DAVFP. The majority of DAVFP collections were made by rangers and research scientists affiliated with the now defunct Canadian Forest Service Forest Insect and Disease Survey (FIDS), over 50 years of regular monitoring for forest pests throughout the Pacific and Yukon Region. The PFC still maintains a strong commitment to maintain and expand the current collections, with greater emphasis on fungal biodiversity research in recent years.

Until a few years ago, records of herbarium holdings were documented in 3 ways: labels on the specimens; hand-written accession books used to assign an individual number to each new specimen; and filed FIDS data forms. To search or examine these records was time-consuming, and required either a written request or a visit to the herbarium. Several years ago, the DAVFP staff began the slow process of converting herbarium records to an electronic format. Some of the data were downloaded from a national FIDS database and reformatted, while the remainder were converted from

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One
B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

written records to spreadsheets. In 1998, Dr. Alan Thomson, of the Pacific Forestry Centre, and a student, Jonathan Bahl, converted these massive spreadsheets into the Herbarium Voucher Specimen Database, available at http://www.pfc.cfs.nrcan.gc.ca/biodiversity/herbarium.

This database currently documents >21,000 of the specimens housed at DAVFP, and may be searched by several different parameters. The data are organized under such fields as "Fungus," "Host," "Determiner" (person making the identification), and "Collector," as well as other categories. Selecting the Web page radio button designating the first letter of the subject further narrows these fields. This executes a search of the database, which returns an updated short-list of names to choose from. One of these names may then be selected and the form submitted. This will return all of the records available that meet the search criteria. Currently, synonyms are not linked, so the searcher should be aware of the taxonomic and nomenclatural history of the species of interest. If there is geographic (UTM, latitude/longitude) information available for the record, the searcher can select a link, which will draw a map showing the collection location. A distribution map of all of the collection locations in a particular search may also be generated, by scrolling to the end of the list and selecting the "submit" button there. Searches may also be conducted on "Locality" (nearest town or drainage division), DAVFP number, or year of collection. Systematists, naturalists, foresters, and other interested individuals can use this database to rapidly determine which species of fungi are housed in the herbarium. A specimen collection slip and collection protocols may also be downloaded from this web site and used by those wishing to contribute samples to the herbarium.

BRITISH COLUMBIA HOST-FUNGUS INDEX

A second database, the British Columbia Host-Fungus Index, was compiled from 60 years of Canadian Forest Service records, augmented from published literature records and additional data provided by Agriculture/Agrifood Canada. The Host-Fungus Index has been designed to assist in the identification and formal documentation of fungi occurring on native plants in the province of British Columbia, with the exception of most agricultural crops and horticultural ornamental (nonnative) plant species. It is organized in a manner similar to the Voucher Specimen Database. There are various plant host and fungus data categories, including, "Genus/Specific Epithet," and "Specific Epithet/Genus." The search by "Specific Epithet/Genus" is recommended for trying to determine the currently accepted genus for a given taxon. This function is useful when the searcher is working with a taxonomic group that has undergone recent revisions. Synonyms are linked to currently accepted names in this database. Two Canadian compendia (Conners 1967, Ginns 1986) are cited for many of the "literature records" herein. They in turn lead to further citations and information regarding voucher specimen locations in herbaria other than DAVFP. The revised format of the Index, and taxonomic modernization is loosely based upon Farr et al. (1989).

BRITISH COLUMBIA MACROFUNGI

Although it might be argued that the data described above merely "scratch the surface" of the true fungal biodiversity in British Columbia, these databases can be used to check records pertaining to some potentially rare macrofungi in the province. "Macrofungi" herein refers to an artificial grouping of fungus species producing fruiting bodies >1 cm in diameter. Macrofungi are more likely to be observed by amateur naturalists or trained parataxonomists, and can often be recognized to genus or species by the collector in the field. For this reason they may be good candidates for preliminary biodiversity studies.

Redhead (1997) lists 65 species of rare or notable macrofungi found in British Columbia, and has assigned preliminary status designations for many of these species, based upon the same designations used by the Conservation Data Centre of the British Columbia Ministry of Environment, Lands and Parks. Most of the species on the list are fleshy fungi in the order Agaricales, commonly known as mushrooms. The reason for this bias is that the majority of the pioneer studies in fungal biodiversity have been, or are currently, conducted on agarics. The taxonomic experts providing verification of this list, including Dr. Redhead himself, reside elsewhere in Canada. Consequently most of the voucher specimens for these records reside elsewhere as well. In fact, a search of the DAVFP database revealed that only 3 of the 65 listed species (*Hypholoma tuberosum*, *Ossicaulis lignatilis*, and *Phaeocollybia kauffmanii*) are represented by collections in this herbarium. This lack of nearby reference specimens makes it difficult for regional taxonomists to confirm the identity of future collections. Research on agaric biodiversity provides much valuable information, as many of these species are intimately associated with indigenous plants as mycorrhizal symbionts or parasites. However, mushroom collection studies are often limited to narrow windows of opportunity during fall fruiting season, which may last a few days or a few weeks at most. The resulting collections must be handled and identified or preserved quickly, and require detailed field notes or photographs. The vagaries of weather from year to year have an impact on the amount and timing of fruiting. These uncertain parameters make it difficult to confirm if a mushroom species is "rare" or if vegetative thalli (mycelia) are in fact common even though the species rarely fruits.

Other groups of macrofungi that are easier to collect,

identify, and observe in the field might be good candidates for determination of "rare" status. Wood-decay macrofungi (Basidiomycota, Aphyllophorales, Polyporaceae) producing woody "conks" on trees or downed wood are the example chosen here. These fungi are large, often showy, and relatively easily identified by naturalists and parataxonomists. Many heart-rot fungi are somewhat host-specific and do not produce significant decay columns and fruiting bodies in trees younger than 80 years. Therefore the search for targeted species is narrowed down to locations where mature tree hosts are located. Many polypores are perennial, often decades old, so they may be observed and collected year-round. This means that seasonal and yearly fluctuations have less of an effect on distribution and incidence data. For example, at DAVFP, 1 conk of *Fomitopsis officinalis* (a perennial polypore causing brown cubical heart rot of mature conifers) has been aged to 53 years. This particular species has a similar habitat, size, and age as the famed "fuzzy sandozi" (*Bridgeoporus nobilissimus*), a huge, rare polypore known only from a few sites in Oregon and Washington, and the first fungus to be listed as an endangered species in the United States (Burdsall et al. 1996). Some polypore species have ethnomycological significance (Blanchette et al. 1992, Blanchette 1997), as well as being essential for tree utilization by some cavity nesting wildlife (Harestad and Keisker 1989). There are also a few rare wood-decay ascomycetes (Xylariaceae) that persist for 1 to 2 years in the field on recently downed deciduous trees, and that appear to be rarely encountered and limited in distribution. These species also warrant further study in order to confirm their rarity.

One of the characters used as an indicator of "critically imperilled" status is: "5 or fewer extant occurrences or very few remaining individuals" (Redhead 1997). However, according to the DAVFP database (Table 1), >1,400 fungus species are represented by a single collection! A total of 2,349 species (67% of all species) are represented by 5 or fewer collections in DAVFP. There is thus an obvious need to separate truly rare species from those that have merely been undercollected or understudied, due to cryptic habit or lack of local taxonomic expertise.

Potential Candidates for Further Distribution Study

With these characteristics in mind, 5 macrofungi have been chosen as potential candidates for further distribution study. The first 3 are polypores, while the other 2 are ascomycetes; all are infrequently encountered despite their size and easily recognizable features. The polypore taxa are described and illustrated in the easily obtained publication by Allen et al. (1996). The species are as follows:

Table 1. Proportions of species represented by 1 or more collections in the DAVFP herbarium of the Pacific Forestry Centre, Victoria, BC.

No. of species	No. of collections/species
1,469	1
436	2
218	3
132	4
104	5
75	6
71	7
51	8
38	9
36	10
68	20
26	50
41	99
2	300
2	400

Fomitopsis officinalis

This large perennial polypore is hoof-shaped, becoming pendulous and cylindrical with age, with old (>50 yr old) specimens reaching up to 40 cm in diameter. The conks are a distinctive chalky white to tan, and often decades old, with a new pore layer developing each growing season. They are most commonly encountered on trunks of mature Douglas-fir (*Pseudotsuga menziesii*) and Sitka spruce (*Picea sitchensis*) on the west coast of the province, but may be found elsewhere on larch (*Larix occidentalis*), grand fir (*Abies grandis*), and ponderosa pine (*Pinus ponderosa*). There are 109 records of *F. officinalis* in British Columbia since 1940 (when data collection first began), but only 19 of those records were added in the past 20 years. It remains to be determined if the reduced sightings and collections are a result of habitat loss (reduction of old growth sites), but the fungus would be a good candidate for further study.

Haploporus odorus

This perennial polypore occurs north of 52°N on willow (*Salix* spp.), and is hoof-shaped with a white lower pore layer and a black pileus. It has a distinct sweet anise odour that can be detected from a distance of several metres. There are only 14 records from British Columbia since 1940, with 5 added in the last 20 years.

Inonotus obliquus

This perennial fungus does not produce a pore layer until its birch (*Betula papyrifera*) host dies, but the large, swollen, orange-brown "blind conks" may be present for many years before this occurs. There are 20 records from British Columbia, with only 4 added in the past 20 years.

Biscogniauxia mediterranea
There are only 3 confirmed herbarium records of this ascomycete, all from southern Vancouver Island and within the past 5 years. However, the fungus produces distinctive, coal-like, flat fruiting bodies, which may extend >7 cm in length, on alder (*Alnus* spp.) and native *Prunus*. It is easily recognized in the field, so lack of earlier collections by regional mycologists might indicate rarity.

Biscogniauxia bartholomaei
This ascomycete forms large numbers of carbonaceous discs breaking through bark of recently fallen red alder (*Alnus rubra*). There is only 1 known Canadian record, which is from southern Vancouver Island. This record is the second North American report since 1911, when it was described by Peck based on a Washington State collection. Worldwide it is only known from 1 other location, the Kamtchatka peninsula, where it was collected on *Alnus kamschatia* (Vasileva 1988). Based on the fact that it has only been collected worldwide 3 times, this fungus would be ranked as G1 by the British Columbia Conservation Data Centre. European monographs of this genus do not include this species.

There are doubtless many other species that are also perennial, fruit on large woody hosts, and are easily recognized in the field by mycologists, parataxonomists, and other naturalists. These fungi would be excellent candidates for monitoring programs in biodiversity studies in provincial parks.

ACKNOWLEDGEMENTS

Past field technicians of the Forest Insect and Disease Survey contributed thousands of records to the B.C. Host-Fungus database. Former herbarium technician Ms. D. Lowe compiled the last printed form of the B.C. Host-Fungus Index (Lowe 1977), and maintained lists of revisions and additions until her retirement in 1992. Ms. F. Ring and Ms. A. Fernando (term herbarium technicians, PFC) undertook extensive reformatting and additional data entry after 1992. Dr. S. Redhead (Agriculture/Agrifood Canada, Ottawa) provided additional database records that have been incorporated into this index. The Canadian Forest Service database, INFOBASE, was used to transfer the majority of the collection data to a spreadsheet used to create the DAVFP Voucher Specimen Database. Dr. A. Thomson, research scientist at PFC, and Mr. J. Bahl, University of Victoria co-op student, provided indispensable advice and technical expertise to the author, setting up the Web page and programming the associated databases.

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Classification and Management of Rare Ecosystems in British Columbia

Donald S. McLennan
Oikos Ecological Services Ltd.
3855 Second Avenue, Box 985, Smithers, BC, V0J 2N0, Canada
Donald@oikos.ca

Irene E. Ronalds
Oikos Ecological Services Ltd.
3855 Second Avenue, Box 985, Smithers, BC, V0J 2N0, Canada

ABSTRACT

This paper reports on the progress in developing a field reference for the identification, description, and assessment of rare ecosystems in British Columbia for the Conservation Data Centre (CDC). To this end, a review of plant community classification concepts is provided. Concepts discussed include (1) the relationship between the plant community and the plant association, (2) vegetation classification concepts used in the biogeoclimatic ecological classification system, (3) clarification of classification concepts and the B.C. CDC's Red- and Blue-listed plant communities, and (4) plant community rarity—the nature of the threat. Management implications and recommendations are outlined as a starting point for further discussion.

Key words: biodiversity, biogeoclimatic ecological classification, protected areas, rare ecosystems.

The objective of integrating conservation of biodiversity as a component of land use strategies has become a common priority in many jurisdictions in North America and around the world. The British Columbia Conservation Data Centre (CDC) was initiated by the British Columbia Ministry of Environment, Lands and Parks in 1991 to collect and standardize information on the status and occurrence of provincially rare vascular plants, vertebrates, and plant communities (Harcombe et al. 1994). Plant communities, as represented by their plant associations, are ranked according to both their global and provincial status using internationally consistent criteria developed by the Science Division of the Nature Conservancy (U.S.). The provincial status is based primarily on the number and areal extent of known occurrences of an element, informed by other factors including abundance, range, protection, trends, and threats (Table 1).

Based on the CDC status ranking, the CDC has identified a preliminary list of 241 rare plant associations for Red- or Blue-listing, according to the criteria outlined in Table 2. Without rare ecosystem inventory and mapping over the whole province, the criteria for Red- and Blue-listing are difficult to apply to rare plant communities at this time. Instead, the CDC uses a "ranking by inspection" system, which is based on available occurrence information, and on the expert opinion of local ecologists, regional ecologists from the Ministry of Forests and the Ministry of Environment, Lands and Parks, as well as experienced naturalists (B.C. Conservation Data Centre 1992).

Information reported in this paper is based on 2 projects carried out by our company for the CDC. These reports identify and map rare ecosystems (Oikos Ecological Services 1998), and summarize floristic and environmental ecosystem characteristics for ecosystems presently included in Red and Blue Lists (Oikos Ecological Services 1999a). This paper summarizes many of our findings from completing these projects, outlines difficulties with rare ecosystem classification and management, and provides our perspective on future directions of the rare ecosystem program.

ISSUES WITH CLASSIFICATION OF RARE PLANT ASSOCIATIONS

Plant Communities, Plant Associations, and Forest Ecosystems

While it is accepted that plant species are distributed individually as populations along environmental gradients, the ecophysiological adaptations of certain groups of plants seem similar, so that their distributions along environmental gradients often coincide. Groups of plants that tend to occur together under similar environmental conditions are called *plant communities*. The distinctiveness of plant communities in nature is often a function of the rate of change of environmental gradients. Where environmental gradients change rapidly, such as at the boundary between a floodplain and a rock cliff, or at the margin of a lake, changes in the plant community are rapid and distinct. Often, however, the

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1 Conservation Data Centre ranks and definitions for element rarity.

1. Critically imperilled	extremely rare (≤5 extant occurrences or very few remaining individuals) or some factor(s) make it especially vulnerable to extirpation or extinction.
2. Imperilled	rare (typically 6–20 occurrences or few remaining individuals) or some factor(s) make it vulnerable to extirpation or extinction.
3. Rare or uncommon	typically 21–100 occurrences; may be susceptible to large-scale disturbances, such as loss of extensive peripheral populations.
4. Frequent to common	>100 occurrences; apparently secure but may have a restricted distribution or future threats may be perceived.
5. Common to very common	demonstrably secure and essentially ineradicable under present conditions.

environmental gradient is more gradual, such as on long slopes in mountainous terrain. In these cases, identifying where one plant community changes to another is much more difficult.

Whereas the plant community refers to real assemblages of plants in nature, the *plant association* is a classification concept that groups similar plant communities, often for environmental description and ecosystem-based land management. In the Braun-Blanquet approach, which is the basis of the plant community classification in the biogeoclimatic ecosystem classification (BEC), plant communities are sampled using relévé analysis, and the floristic data are analyzed so that similar plant communities are grouped together, following the rules of vegetation classification (see Mueller-Dombois and Ellenberg [1974] or Shimwell [1971]). Using this approach plant communities are considered part of the same plant association if they meet certain criteria for frequency of occurrence and percent cover of indicator species. The CDC has recently updated its terminology so that rare plant associations, rather than rare plant communities, are the elements of conservation (B.C. Conservation Data Centre 1999).

Ecosystems can be defined as the system that includes the total assemblage of living organisms, together with their nonliving environment, over some particular area (Kimmins 1992). The nonliving component of the ecosystem includes factors such as climate, physiography, and soil, and are together considered as *site* component of the ecosystem. The plant community is the plant component and, with animals, makes up the biotic component of the ecosystem. Thus *plant community* and *ecosystem* should not be considered synonyms.

Table 2. B.C. Conservation Data Centre criteria for Red- and Blue-listed plant communities.

Status	Significance	CDC rank
Red-listed	endangered or threatened	plant communities with ranks of 1, 2, 1–2, 1–3
Blue-listed	vulnerable and at risk	plant communities with ranks of 2–3, 3, or 3–4

The size of an ecosystem is not included in the definition, and is determined by scale of interest. In the BEC system local forest ecosystems are defined as areas of the landscape uniform in soil polypedon, vegetation composition, and vegetation structure (Pojar et al. 1987). These are also the criteria used when selecting the location for a plot to conduct releve sampling of a plant community. Rare plant associations are thus used as labels for defining and identifying rare ecosystems that include identified rare plant communities. The rare ecosystems that support listed rare plant communities should be the focus of conservation to ensure the coarse filter approach to conservation of biodiversity.

Applications of the Plant Association Concept in Biogeoclimatic Ecosystem Classification

The concept of the plant association is central to the classification of ecosystems in BEC at the chronological (successional), regional (subzone), and local (site association) levels of the system (Pojar et al. 1987). Within BEC, changes in plant communities and their plant associations across the landscape are used as bio-indicators to interpret important ecological gradients, and for determining the location of important ecological boundaries. For example, plant communities change over time on the same site as vegetation succession proceeds following stand replacing disturbance. Forest stands develop in a continuous and relatively predictable manner that has been classified into a number of *structural stages* (Initial, Herb, Shrub-Herb, Pole Sapling, Young Forest, Mature Forest, and Old Forest, following RIC [1998]). Plant communities, and the plant associations that circumscribe them, also change along disturbance chronosequences, and floristically unique plant associations along this time gradient are referred to as *seral associations*. A structural stage may have many seral associations within, especially in the Initial, Herb, and Shrub-Herb stages, or the same seral association may include more than one structural stage. This latter situation occurs primarily in midsuccessional forest structural stages, such as the Pole Sapling and Young Forest stages, where subcanopy plant species diversity is often low.

Another important application of the plant association

concept in BEC is used to identify biologically important climatic and edatopic boundaries. The distribution of distinct plant associations that occur in Mature Forest and Old Forest structural stages on zonal sites marks the geographic extent of regional climates, or *biogeoclimatic subzones*. Within subzones, the distribution of plant associations that occur on Mature Forest and Old Forest structural stages on nonzonal sites marks the geographic extent of many different *site series*. Thus, a site series is a group of sites that are climatically uniform, and that have the same ecological potential, as indicated by the plant association that characterizes it in Mature Forest and Old Forest structural stages. The same plant association may occur on azonal sites in more than 1 subzone; this group of sites is referred to as a *site association*. Site associations and site series may occur on a range of *site types* where compensating factors result in ecological equivalence, as expressed by the Mature Forest and Old Forest plant associations that characterize the site association. Thus any 1 rare plant community may occur on a wide range of rare ecosystems, and is defined by the rare plant association listed by the CDC.

Using the Plant Association Concept for Rare Plant Associations

The CDC lists rare plant associations in British Columbia to identify ecosystems considered to be at risk. Many of the listed plant associations are the plant associations of forest ecosystems in the Mature Forest and Old Forest structural stages that are used in BEC to identify and define site associations. These older structural stages are rare or threatened across the landscape, and are generally the object of conservation. Younger structural stages that have developed following forest harvesting or other disturbance within these site associations are represented by different plant associations (seral associations) and, generally, are not considered rare or endangered.

We have stated that the plant association is used to classify plant communities by sampling plant communities across the landscape in a standardized manner, and then using the methods of phytosociological analysis to group similar plant communities into plant associations. The rare plant association acts as a label to describe all rare plant communities (and rare ecosystems) that have a similar floristic composition. The floristic composition of the rare plant association is then used as a criterion to identify rare plant communities in the field. Potential rare plant communities are sampled using the releve sampling procedure and membership in the rare plant association is determined by their floristic similarity.

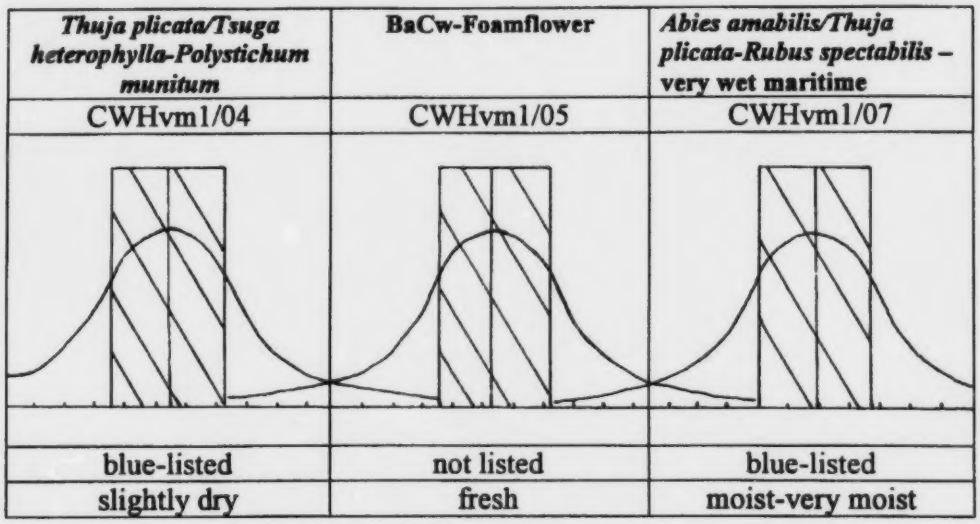
At present, no criteria have been established for determining how similar the sampled plant community must be to the target rare plant association to consider it as an example of that rare plant association. A potential situation for the Submontane Very Wet Maritime CWH Variant (CWHvm1) is illustrated in Figure. 1. The *Thuja plicata/Tsuga heterophylla - Polystichum munitum* and *Abies amabilis/Thuja plicata - Rubus spectabilis* - very wet maritime rare plant associations are separated by an unlisted plant association, BaCw - Foamflower. The bell curves are intended to show the central concept (or average condition) of each of the 3 plant associations, and how the floristic composition of 30 plant communities varies along the underlying moisture gradient. The critical question is, "At what point along this moisture and similarity gradient do we assign membership to the 2 rare plant associations?" The gradient need not be soil moisture or soil nutrients. It may be a gradient of increasing herbivory in a grassland ecosystem, or increasing invasion of nonnative plant species near developed areas that is causing the floristic variation from the central concept of the rare plant association. To successfully apply the plant association concept to the conservation of rare ecosystems similarity criteria must be developed to determine if sampled plant communities belong to the listed plant association. These criteria could be derived by multivariate methods such as discriminant analysis and built into existing vegetation tabulation software such as VENUS (B.C. Ministry of Environment, Lands and Parks and B.C. Ministry of Forests 1997) or VTAB (Kayahara 1992).

An additional problem with identifying rare plant community membership is the lack of data for many of the listed associations. Use of similarity indices will require solid databases to define the average floristic composition of rare plant associations for comparison with candidate plant communities sampled in the field. At least 5 sample plots are required for reliable vegetation classification and as many as 10 are desirable. At this time, many of the rare plant associations listed by the CDC are defined by fewer than 5 plots, and sometimes by only 1 or 2. A low number of sample plots will make the application of multivariate methods for deciding membership in a rare plant association unreliable. More field sampling is required for rare plant associations that are defined by less than 5 plots.

CONSERVATION OF RARE ECOSYSTEMS

Relationship to the Biogeoclimatic Ecosystem Classification

Rare plant associations and the ecosystems that they define are largely based on information collected for developing the BEC. This integration with BEC facilitates efforts to conserve rare ecosystems, because BEC provides an ecological framework for identifying, mapping, modelling and managing rare ecosystems. For example, terrestrial ecosystem mapping (TEM) uses BEC, and has recently been carried out in many tree farm licences (TFLs) and other forest licence areas in British Columbia, TEM provides an immediate inventory of rare ecosystems in the areas mapped (Fig. 2). The identification and mapping of rare



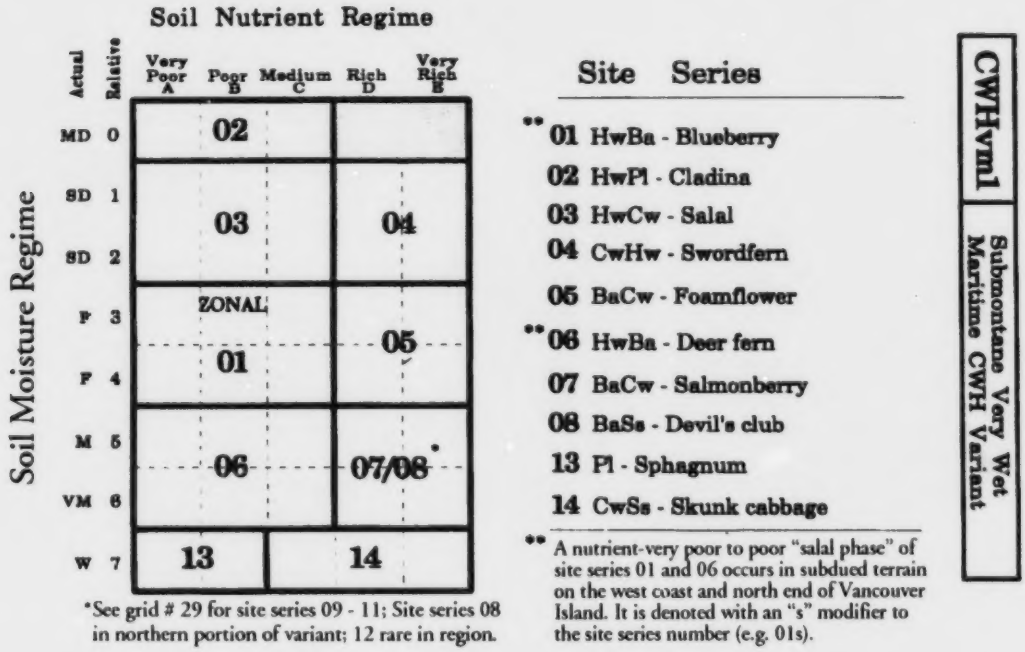


Figure 1. Changes in floristic similarity of 30 plant communities across 3 site series along a moisture gradient in the CWHvm1 variant. Site series in the upper diagram can be located on the edatopic grid in the lower diagram (grid from Green and Klinka 1994).

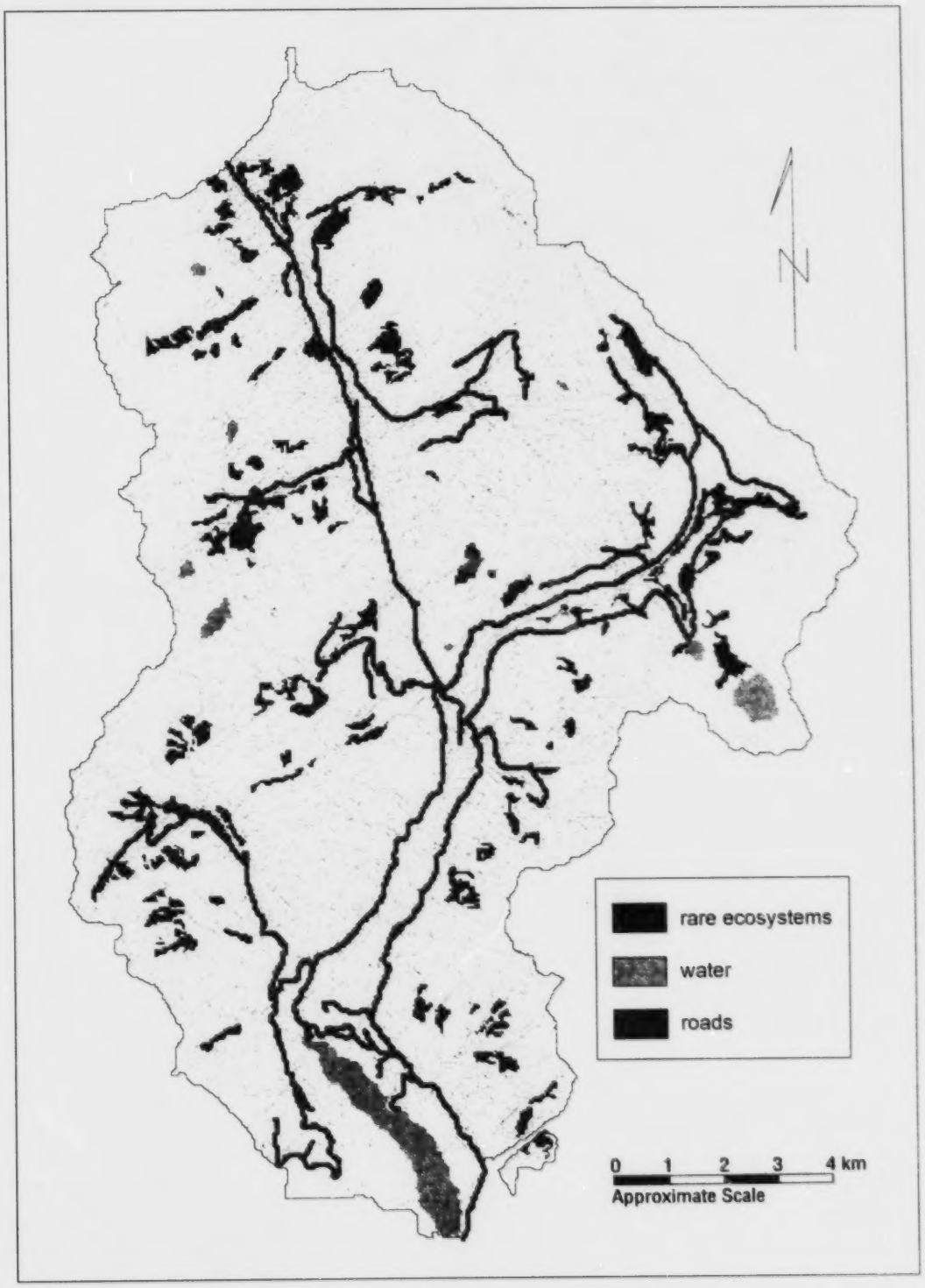


Figure 2. Distribution of rare ecosystems in the Capilano watershed (Oikos Ecological Services 1999*b*).

ecosystems in operational areas makes it possible to account for them in landscape unit planning and harvesting plans. TEM uses a standardized approach to defining stand structural stages so that the long-term recovery of the Mature Forest and Old Forest stages that define many rare plant associations can be predicted (Fig. 3).

Rare Ecosystems in Protected Areas

Rare ecosystems in British Columbia occur on public land in protected areas and in operational forest and mining areas, as well as on private land. Because a rare ecosystem occurs in a protected area does not mean it will be conserved. For example, in BC Parks there is a movement towards allowing natural disturbances such as fire and pest outbreaks to create a natural mosaic of ecosystems. Under this regime rare ecosystems will be disturbed along with those that are not listed, and this may significantly decrease rare ecosystem representation. Conversely, fire suppression may also result in undesirable changes to the plant community, such as forest ingrowth. A significant part of the problem is the lack of inventory on rare ecosystem occurrence and distribution in our protected areas. Without this information it is difficult to decide on the best policies for managing protected areas, and the rare ecosystems they contain.

Rare Ecosystems in Operational Areas

Rare ecosystems are also widespread on public lands where forest harvesting and other kinds of development are planned. At this time only 4 of 241 rare ecosystems are listed in the Identified Wildlife Management Strategy (B.C. Ministry of Environment, Lands and Parks and B.C. Ministry of Forests 1999). The remaining 237 rare ecosystems do not have official protected status and this situation should be rectified as soon as possible. Note that just listing the rare ecosystems provides an important service in identifying them, because land managers are aware of their existence and may consider them in development plans.

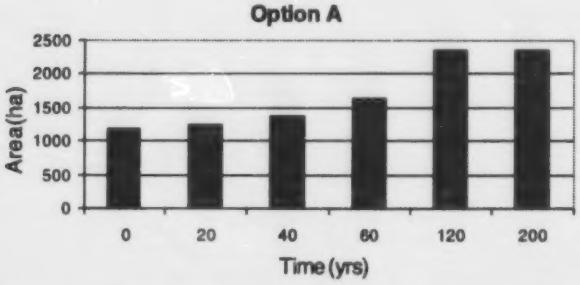


Figure 3. Changes in the area of CDC rare ecosystems over 200 years in the Capilano watershed (Oikos Ecological Services 1999b).

Without a complete ecosystem inventory for all areas presently being developed, rare ecosystems should be identified as part of the development process. Ideally, rare ecosystems should be identified as part of the landscape unit plan so that harvesting plans can optimize rare ecosystem conservation and resource extraction. If not identified at the landscape level before development, rare ecosystems can be identified at the silviculture prescription stage. To accomplish this effectively, materials and training sessions will need to be developed that facilitate the identification of rare ecosystems. Field personnel conducting silviculture prescriptions are already trained in the identification of BEC site series, so they could easily learn to identify the Mature Forest and Old Forest stands within site series where rare ecosystems occur. Once identified, whether at the landscape unit plan stage or at the silviculture prescription stage, the locations of rare ecosystems can be noted on forest cover maps and noted permanently as the maps are updated.

Ecosystem-Level Conservation Strategies for Rare Ecosystems

Once they have been identified and located, rare ecosystems will require a management plan to ensure their continued protection. Conservation of rare ecosystems at the landscape level is an important component of rare ecosystem conservation and approaches are well summarized by Noss (1995). However, our paper deals with the development of a rare ecosystem management approach at the local level, where individual ecosystems have been identified for protection.

A management plan for local rare ecosystems seeks to conserve ecosystem structure, composition, and function. This objective is best met by conserving ecosystem processes that support these ecosystem characteristics. Another aspect of BEC relevant to effective conservation is the analysis of an ecosystem process that is inherent in ecosystem interpretation. Processes important to maintaining ecosystem integrity may occur completely within or outside the ecosystem boundary. Examples of internal processes to be conserved are mineral soil and humus form processes (e.g., mineral weathering, moisture and nutrient uptake, and mineralization of organic matter), and stand structural dynamics (e.g., dead wood cycling, microclimatic effects, reproduction, and plant interactions). External processes include hydrologic effects such as the seasonality and chemical composition of seepage water, water table depths, or the periodicity of flooding that characterizes the ecosystem. Other important processes determined by factors outside the ecosystem are climatic conditions that determine natural fire frequency and periodic windthrow events, and mesoscale factors such as cold air drainage and deep snow accumulation.

Effective rare ecosystem management plans will include recommendations for preserving ecosystem structure,

composition and function, and should consider maintenance of internal and external ecosystem processes. Internal ecosystem characteristics and processes can be protected by establishing a rare ecosystem management and/or reserve zone around the ecosystem, and by excluding all extractive and development activities. Recreational use should be restricted or carefully planned to prevent ecosystem degradation. The width and other details of management/reserve zones around rare ecosystems will vary depending on the ecosystem structural attributes that require protection. For example, in forested ecosystems, the management and reserve zones should be designed to protect the structure of the ecosystem by reducing windthrow risk. It may sometimes be desirable to reduce fuel continuity in these reserves to decrease the risk of fire damage. Reserves around protected grassland or shrub-steppe ecosystems should be designed to reduce border effects of grazing, including trampling and soil disturbance that may provide access for nonnative species. Often, fencing will be required to protect grassland and shrub-steppe ecosystems from overgrazing, especially on private land or public range areas.

Protecting ecosystem processes that are external to the boundary of the rare ecosystem will be much more difficult than the protection of internal processes, and will sometimes be impossible. Many rare ecosystems depend on hydrologic effects that are determined by factors outside the ecosystem. For example, wetlands occur as complex ecosystem mosaics where each wetland ecosystem is controlled by often slight differences in hydrologic regime. The negative impact of road building on wetland hydrologic regime, and consequently on wetland ecosystem structure, composition, and function, is easy to observe along all of our highways and logging roads. Many rare forested ecosystems are located on floodplains and alluvial fans, where above- and below-ground flooding are important determinants of ecosystem function. These hydrologic factors should be considered in management plans designed to protect the rare ecosystems that rely on them. Recommendations for upslope road building and drainage, area-based cutting restrictions, and hydroelectric development should accompany rare ecosystem management plans to ensure maintenance of these ecosystem processes.

In some rare ecosystems, periodic fire is essential for maintenance of ecosystem structure and composition; however, fire control has permitted the establishment of shrubs and poorly growing trees that would not occur under a natural fire regime (Haeussler and Heatherington 2000). Many grassland and shrub-steppe ecosystems throughout the province will require prescribed burning for their maintenance, and optimal fire periodicity and intensity for these ecosystems should be determined. Other external factors such as maintenance of cold air drainage, snow accumulation areas in alpine ecosystems, or significant slope effects in semiarid climates are beyond the scope of management, but in general should be little affected by development.

MANAGEMENT IMPLICATIONS AND RECOMMENDATIONS

The rare ecosystem list produced by the CDC represents an important starting point for protection of biodiversity in British Columbia. The list should be seen as the beginning of a process—much has yet to be done. At present 186 of the 241 plant associations listed are Mature Forest and Old Forest stages of overharvested site series. The remainder include shrub-grasslands and grasslands in dry subzones, deciduous stands on floodplains, forested and nonforested wetlands, alpine ecosystems, and marine foreshore ecosystems. Wetlands and alpine ecosystems are clearly underrepresented, and the recent wetland classification being completed for British Columbia will help fill in this important area (see MacKenzie and Shaw 2000). Alpine ecosystems require more work but the priority is relatively less given reduced pressure on alpine ecosystems.

To continue protecting rare ecosystems in British Columbia, several areas have a high priority for action.

1. Rare ecosystem protection needs to be integrated into the operational processes of forest harvesting, mining, and other land development as soon as possible by providing training and distributing field materials so that technical field staff and government and industry planners can account for rare ecosystems in their operational activities. Rare ecosystems in all completed TEM areas should be identified as a theme and included in development plans. Inclusion of all listed rare ecosystems in a revised Identified Wildlife Management Strategy should be a priority, and the full implementation of the Forest Practices Code *Biodiversity Guidebook* (B.C. Ministry of Environment, Lands and Parks and B.C. Ministry of Forests 1996) at the site level should be reinstated.
2. Floristic similarity criteria need to be developed for determining plant community membership in rare plant associations. To reliably apply these criteria, vegetation databases for the present list of rare ecosystems should be upgraded so that each rare plant association is defined by a minimum of 5 plots, and ideally 10 plots.
3. The present list of rare ecosystems should be upgraded and studied in each forest region to confirm the present designations, to exclude those rare ecosystems not truly requiring protection, and to add new ecosystems as required.
4. Management approaches for protecting rare ecosystems at the landscape and local ecosystem levels should be developed and published so that they can be included as operational activities. Landscape-level protection should follow an approach similar to that proposed by Noss (1995). At

the local level, where relatively isolated rare ecosystems are excluded from harvesting or other development, research should be conducted to determine the size and nature of management and reserve zones for different types of rare ecosystems. Details of prescribed fire for rare ecosystems requiring periodic fire for their maintenance should also be researched.

5. To ensure a comprehensive approach to conservation of rare ecosystems, surveys of rare ecosystems in protected areas should be surveyed to provide critical information on rare ecosystems that are presently protected from development. Based on these inventories management plans should be developed for rare ecosystems within protected areas.

British Columbia is fortunate enough to have landscapes and ecosystems that are important both regionally and globally; we should consider it our public responsibility to preserve the integrity of all components of our natural biodiversity mosaic. The CDC listing of rare ecosystems is an important contribution to this objective, but identification and protection of these important ecosystems are incomplete. Completing the identification and classification, and implementing effective rare ecosystem policies should be a high priority for all land managers in British Columbia.

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Invertebrates from Caves on Vancouver Island

Patrick Shaw
British Columbia Speleological Federation
792 East 15th Avenue, Vancouver, BC, V5T 2R7, Canada
dps@istar.ca

Martin Davis
Island Karst Research
c/o 792 East 15th Avenue, Vancouver, BC, V5T 2R7, Canada

ABSTRACT

Casual observation and collection of invertebrates in caves on Vancouver Island, B.C. have been conducted since the early 1960s. Two comprehensive and systematic surveys have been carried out: one in the 7 caves of Horne Lake Caves Park in 1986; and a second in 1997, which covered 15 caves of a wider geographic area on the island. A total of 192 taxa have been identified to date, including a number of new species, records, or rare species.

Key words: cave, invertebrate, Vancouver Island.

The fauna of caves in Canada is quite poorly known. Until recently, it was presumed that the active glacial history in Canada would have led to the extirpation of any obligate troglobites. As such, the present cave fauna would have immigrated from source areas beyond the southern ice limit.

Collections and discoveries in cave areas in the Rocky Mountains (Holsinger 1980, Bousfield and Holsinger 1981, Zacharada and Pugsley 1988), on Vancouver Island (Holsinger and Shaw 1988), and on Prince of Wales Island, Alaska (Carlson 1995) have proven this not to be the case. However, systematic studies of cave invertebrate communities in Canada have been only rarely conducted (Peck 1988) and the composition of the subterranean fauna in most areas of the country remains relatively unknown. Scudder and Cannings (1994) recognized this lack and identified caves as a priority for invertebrate inventory.

The composition of cave faunas on Vancouver Island is now relatively well studied compared to other karst areas in Canada. In addition to sporadic collections, 2 concerted studies of cave biota have been conducted. The first was a summer/winter survey of 7 caves in Horne Lake Caves Park in 1986. A second survey, conducted during the summer and fall of 1997, covered a variety of caves in 3 areas of the island. These covered a range of lengths, elevations, and settings, but were a tiny subset of the potential locations for study. Some preliminary observations are presented here as an introduction to the biology of west-coast caves, with a more complete assessment in preparation.

METHODS

General locations of the surveyed cave systems are shown in Figure 1. Efforts were made to cover a range of cave size, elevation, and geographic locales, with collections at regular to semi-regular intervals throughout systems. Field collections used a wide variety of techniques. Careful visual searches were conducted with aspirators and brushes. Pitfall traps baited with a variety of attractants and set for 3–10 days were important in the surveys. Aquatic fauna were collected by searching, Surber samplers, and drift net sets. Some surface springs were also sampled.

RESULTS

- A total of 191 taxa have been identified in collections thus far, including a minimum of 10 new species.
- The highest diversity occurs in the entrance area, and the composition is dominated by taxa that would likely be

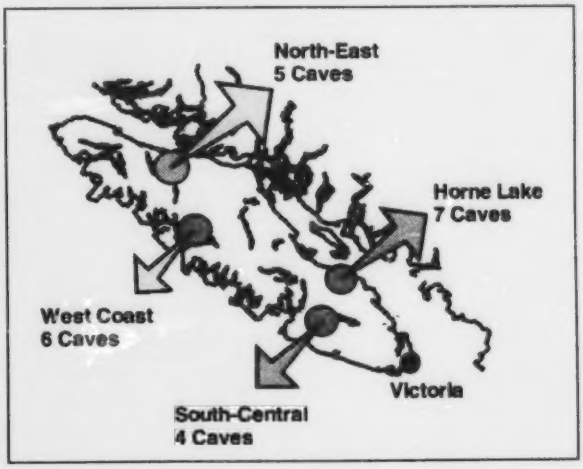

Figure 1. Map of Vancouver Island showing general cave areas sampled for invertebrates to date.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

found elsewhere in similar surface environments with conditions such as low light and high humidity (Fig. 2). The near-entrance fauna was dominated by a variety of Diptera, and associated predators such as Aranae and Coleoptera.

- In deep-cave habitats, collections were dominated by Collembola, Symphyla, and Diplura, with infrequent captures of other taxa, such as Acarina (*Robustocheles occulta*), Diptera, or Siphonaptera.
- At the order and family level, the composition of cave fauna on Vancouver Island is similar to that found elsewhere in North America (Holsinger and Culver 1988). Similarities, even at the species level, between Vancouver Island collections and similar sites in southeast Alaska are particularly remarkable.
- Few endemic troglobites have been found to date in the collections; however, many taxa are currently identified to genus or higher levels, and closer examination may yield further discoveries. This is particularly true of the Diptera, for which new species are certain, even in the present collections.
- Of particular interest in the collections have been the Crustacea, with 2 certain troglobites known.

Composition of Collections

The general composition of the collections is shown in Figure 3.

Coleoptera

Thirty-seven species have been identified in the collections. The majority (19 species) belonged to the Staphylinidae, a family of mostly predatory beetles. The next largest group (10 species) are the ground beetles (Carabidae), common in forest habitats in moist, dark places.

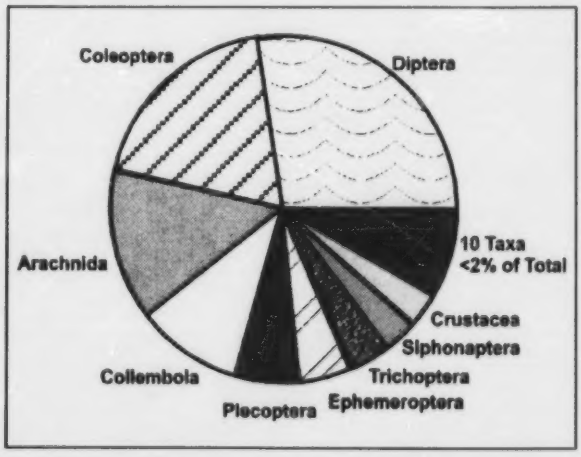

Figure 2. Overall composition of taxa collected during cave invertebrate surveys on Vancouver Island.

Arachnida

The collections include 17 species of Aranae, 10 opilionid species, and 1 identified predatory acarine (*Robustocheles occulta*). Nearly all are probable troglophiles/ trogloxenes. One common opilionid (*Nelima paessleri*) overwinters in dense hibernating-aggregations in mines and caves (Holmberg et al. 1984). The acarine diversity in the collection is grossly underestimated, since none of the aquatic taxa have yet been identified.

Collembola

Twenty taxa, many still to be identified to species. Collembola are the most common component of the deep-cave habitats. Several may be obligate troglobites, species of the genus *Arrhopalites* in particular.

Plecoptera/Ephemeroptera/Trichoptera

These orders were represented by a total of 25 species, mostly

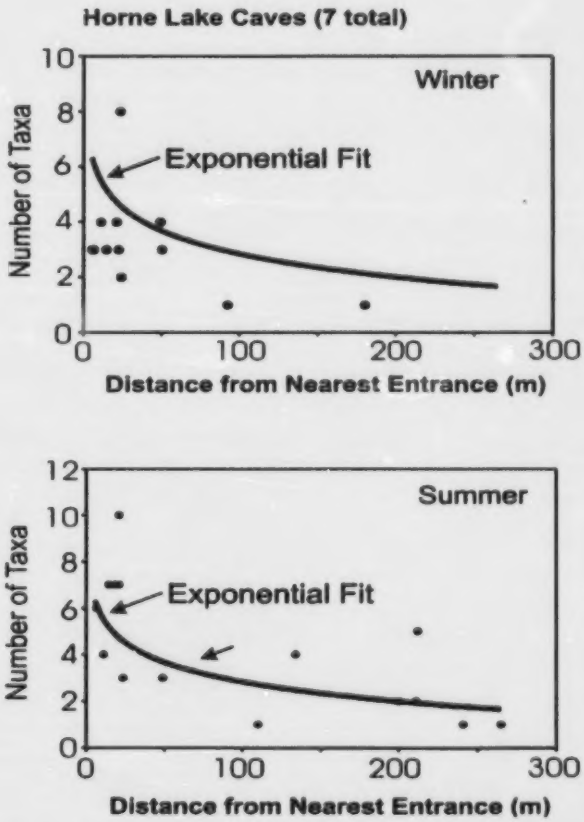

Figure 3. Comparison of summer (August–September) and winter (February) pitfall collections in 7 caves in Horne Lake Caves Provincial Park, Vancouver Island.

as aquatic larvae. All of the "EPT" taxa are probably "accidentals," washed into the cave through surface streams.

Siphonaptera

Seven species have been collected underground. Most are parasites of wood mice (*Peromyscus* spp.), which make extended excursions into caves and were attracted to pitfall trap sets. One species was collected from *Myotis* spp. during bat studies.

Crustacea

Two troglobites: the amphipod *Stygobromus quatsinensis* (Holsinger and Shaw 1988); and an undescribed bathynellid of the genus *Pacificabathynella*. Collections have shown that *S. quasinensis* is widely distributed on Vancouver Island and, surprisingly, has been recently collected from cave habitats in the offshore islands of southeast Alaska (Carlson 1995). In addition, there are several probable new copepod species in the genus *Acanthocyclops*.

Diptera

Fifty taxa have been collected, many of which have as yet been identified only to genus. In the collections is a particularly strong dominance (19 taxa) of Mycetophilidae and closely related families. One mycetophilid genus alone (*Speolepta*) is probably represented by 2–3 undescribed species (A. Borkent, Royal British Columbia Museum, 1998, pers. comm.). A key discovery in 1997 was that of an extremely rare simuliid (*Parasimulium meleanderi*; Borkent and Currie in prog.), which probably has a groundwater-dwelling larva. Most species are apparently restricted to areas near the cave entrances, from full light to twilight regions. There are probably few true troglobites, although study of the life-history of some of the mycetophilids would probably yield some important information.

Other Taxa

Amongst the remaining 15 taxa are representatives of 10 orders or phyla. Two common species of Lepidoptera (*Triphosa haesitata*, *Scoliopteryx libatrix*) are represented, as are a species of symphylan, and >3 species of aquatic oligochaetes. At least 4 species of myriapods were collected, including a probable new genus of Conotylidae (W. Shear, Hampden-Sydney College, VA, 1999, pers. comm.).

CLOSING REMARKS AND RECOMMENDATIONS

- These are very preliminary studies and the observations thus far are suggestive of the potential for further discoveries.
- Collections thus far have focused on coastal karst; there needs to be further (or similar) effort in karstic terrains elsewhere in British Columbia and Canada.
- Both high endemicity and extreme scarcity of some fauna collected in caves on Vancouver Island (such as *P. meleanderi*) may be a function of collection effort; additional collections need be done to establish the full geographic extent of many of these species.
- Surface environs in karst areas, particularly in coastal areas, are subject to massive change through timber harvesting. While this probably most strongly affects the entrance zone, particularly with respect to humidity and temperature regimes, the influence may extend deep into the cave through water flow and evapotranspiration changes. Careful environmental and faunal studies of caves before and after timber harvest could assess the effect of these changes.
- There is a surprisingly high faunal similarity between cave taxa collected on Vancouver Island and in similar environs on coastal islands in Alaska. Genetic studies of local populations on Vancouver Island and Alaska (and probably the Queen Charlotte Island) would establish the time of dispersal and provide important information on colonization of offshore islands.

ACKNOWLEDGEMENTS

Special thanks to the many enthusiastic cavers who volunteered both their time and inspiration during field work. A special thanks to A. Vanderberg who was instrumental in the 1997 collections. Their support was and will continue to be invaluable. Many taxonomists contributed their expertise to this work. In particular, Dr. A. Borkent, Dr. J. Klimachewski, Dr. K. Christiansen, and R. Crawford are particularly thanked. Theirs is a seriously undervalued and essential skill in the proper assessment of biodiversity. This work was supported by funds from British Columbia Parks (1986) and from the British Columbia Conservation Data Centre (1997).

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Wildlife and Wildlife Habitat Inventory Planning to Meet Land-Based Program Planning Needs in British Columbia

Dennis A. Demarchi

Wildlife Inventory Section, Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9377 STN PROV GOVT, Victoria, BC, V8W 9M1, Canada

Richard P. Thomas

Wildlife Inventory Section, Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9377 STN PROV GOVT, Victoria, BC, V8W 9M1, Canada

Bruce A. Pendergast

Wildlife Inventory Section, Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9377 STN PROV GOVT, Victoria, BC, V8W 9M1, Canada

ABSTRACT

Wildlife and wildlife habitat inventory in British Columbia is designed for multi-scales of both planning and inventory. Planning processes can vary from international (cooperation on grizzly bear management), provincial (Protected Areas Strategy), regional (Land and Resource Management Planning), subregional, landscape unit planning, or forest development planning, to local (operation resource allocation planning). To accommodate that level of complexity many resource inventories have been standardized through the multi-agency Resources Inventory Committee. Most of the ecosystem-based inventories and classifications currently being recommended in British Columbia are not stand alone; in fact, they accommodate portions of classifications or information from many other inventories.

Key words: inventory, Terrestrial Ecosystem Mapping, Vegetation Resource Inventory, wildlife habitat planning.

Wildlife habitat value is the result of interpreting the ecological or site series information from Terrestrial Ecosystem Mapping (TEM) products or surrogate site series products to assign values for select wildlife species (Resources Inventory Committee 1998*a,b,c,d,e*). Such habitat value may be a measure of the current suitability of the habitat to support a given species' living requirements, or of the potential capability of the habitat in the correct successional stage or under specific management to meet a given wildlife species' living requirements (Resources Inventory Committee 1998*d*).

Wildlife and wildlife habitat inventories in British Columbia provide information about wildlife distribution, status, and abundance, and relative information on the ecology of supporting habitats. Those inventories provide data for ecological maps and for various planning agencies whose resource management impacts on wildlife species' habitat characteristics, distributions, and relative abundance, as well as on ecological data for interpretations of habitat sensitivity, current habitat (suitability), and potential habitat (capability) for wildlife production or support. Intensive inventories generate additional information needed for the designation and management of specific wildlife populations, and species and habitats at risk, and for the development of detailed operational plans, guidelines, and practices to meet the requirements in various provincial and regional resource development planning processes.

Wildlife and wildlife habitat inventory projects must be balanced between scale and level of survey intensity and the planning level in which the inventories will be used. To be useful, inventory projects must also meet the planning horizon of the intended plan, and a prioritized planning schedule for input into various plans (assuming that all planning is being generated from the broad to site-specific levels).

STRATEGIC GOAL

The overall goal of the wildlife species and wildlife habitat inventory program in the Resources Inventory Branch is to provide site-specific inventories of those resources across the province to meet operational and higher level planning requirements.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Wildlife and wildlife habitats occur everywhere in the province; however, only where there are competing uses for resource extraction and management is it necessary to have a complete detailed inventory. The inventory requirements can be considered within various planning levels from the national, through provincial, regional, subregional, landscape, and operational levels (Fig. 1). Just as not all areas of the province will be covered under subregional or Land and Resource Management Planning (LRMP), not every area will have forest development or resource extraction. Therefore, not every area in the province needs the most detailed wildlife and wildlife habitat inventories.

At the national and provincial levels, ecoregion (Demarchi 1995, 1996) and biogeoclimatic zonation classifications (Pojar et al. 1987, Meidinger and Pojar 1991, B.C. Ministry of Forests 1994) provide enough wildlife and wildlife habitat information for resource planning. At the regional and subregional (LRMP) levels, overview ecosystem classifications that incorporate ecosections, biogeoclimatic subzone/variants, and broad ecosystem classes (Resources Inventory Committee 1998*a*) provide the most meaningful information. At the landscape unit level, general ecosystem classifications (Resources Inventory Committee 1998*b*) that provide some detail on landforms, terrain and site series, while incorporating ecosections and biogeoclimatic subzone/variant information, supply meaningful information for resource planning. Finally, at the logging or cutblock level, detailed ecosystem information is required to make sound resource management decisions.

NEEDS AND DRIVERS

Specific business requirements include:

- Information that will enable the Ministry of Forests (MOF) District Managers to be satisfied that an operational plan is both in accordance with the Forest Practices Code (FPC), and that its implementation will adequately manage and conserve the wildlife resources.
- Information about a species at risk and its habitat requirements for inclusion in the *Species and Plant Community Accounts for Identified Wildlife* (B.C. Ministry of Environment, Lands and Parks 1997): information for Wildlife Habitat Area (WHA) establishment at levels of detail suitable for decisions about both where and how to log with minimal negative impact on wildlife species and to optimize positive effects. (A Wildlife Habitat Area [WHA] is a mapped area of land that the Deputy Minister of Environment, Lands and Parks [MELP], or a person authorized by that Deputy Minister, and the Chief Forester, has determined is necessary to meet the habitat requirements of one or more species of identified wildlife.)
- Input to landscape unit plans through the development of wildlife management objectives or the accommodation of "identified wildlife" (animals and plants) in forest development plans, including sensitive area and WHA mapping and preparation and application of species' accounts for the *Species and Plant Community Accounts for Identified Wildlife* (B.C. Ministry of Environment, Lands and Parks 1997).

ELIGIBILITY CRITERIA

The following wildlife and wildlife habitat inventories are critical to meet land-based resource management objectives.

Detailed Wildlife Habitat Identification

This inventory addresses forest licensee responsibilities for the identification and mapping of site series classification (at 1:5,000 or 1:10,000) of habitats for "stand level" operational planning under the FPC. Content includes presence of any of

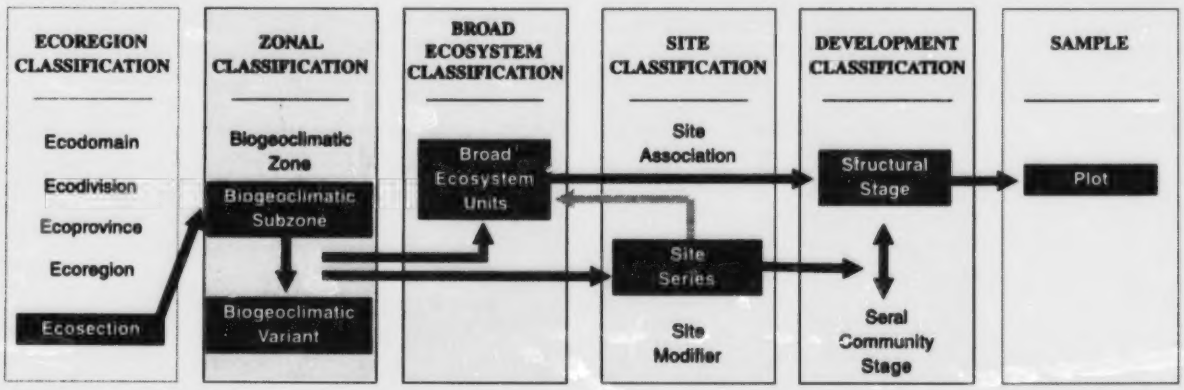

Figure 1. Relationship among the various ecosystem classifications used for land-based resource management in British Columbia (modified from Resources Inventory Committee 1998*b*).

the wildlife species defined in the FPC Operational Planning Regulation, and WHA identification. Coverage is limited to habitats that will be affected by active access developments and forest harvesting.

Wildlife Habitat Inventory and Absolute Abundance Wildlife Inventory (1:20,000)

The sample-based 1:20,000 inventory is required for all forest management areas that within the next 10 years will be undergoing active forest development planning. Detailed 1:20,000 wildlife habitat inventories and absolute abundance wildlife species surveys provide information about the characteristics and distribution of wildlife species and their habitats and about potential locations of WHAs. This level of information contributes to decisions in the early stages of forest development planning about where to log, and to the establishment of landscape-level biodiversity objectives. They also provide the framework for limiting and focusing the detailed habitat requirements in logging plans and silviculture prescriptions.

General-Level Wildlife Habitat Inventory and Relative Abundance Wildlife Species Inventory (1:50,000)

The sample-based 1:50,000 inventory is required for forest management areas having high wildlife values that will not be undergoing forest development within the next 10 years. The 1:50,000 general-level habitat inventories and relative abundance wildlife species surveys provide information about the characteristics and distribution of wildlife species and their habitats and about approximate and potential locations of WHAs. This level of information contributes to decisions in the early stages of landscape-level planning about sensitive habitats and to the establishment of some landscape-level biodiversity objectives. They also provide the framework for limiting and focusing the detailed habitat information requirements in logging plans and silviculture prescriptions.

Wildlife Species and Population Inventories

These inventories further address government's responsibility under the FPC to provide landscape-level biodiversity objectives, to identify and characterize species at risk, and to determine the measures required to protect critical habitats of those species at risk that have been designated as identified wildlife in the *Identified Wildlife Management Strategy*.

Overview Wildlife Habitat Inventory and Presence/Absence Wildlife Species Surveys (1:250,000)

These surveys add field information on wildlife species presence where little is known or documented. They will occasionally be required, primarily for broad area strategic planning and for prioritizing landscape units for more detailed assessments. Overview species surveys provide data for habitat management options over broad areas—more intensive work is used for defining cutblock activities, identifying WHAs, and determining the scope of general wildlife habitat conservation measures within the *Identified Wildlife Management Strategy*, and Wildlife, Biodiversity and Riparian Management Area guidebooks.

STANDARDS AND QUALITY ASSURANCE DELIVERY

Standards have been developed for all wildlife species and habitat inventories through the Resources Inventory Committee. These include standards for data collection (B.C. Ministry of Environment, Lands and Parks and B.C. Ministry of Forests 1998), terminology, classification, mapping, and reporting. Training courses are being developed and delivered to assure competency, to clarify standards, and to support their consistent application.

Quality assurance procedures have been developed for ecosystem field plot measurements, both small- and large-scale terrestrial ecosystem mapping (TEM) wildlife habitat capability and suitability assessments, and wildlife species inventories. These procedures include contract specifications, and pre-field, field, and post-field monitoring. For TEM products, quality assurance is a formalized 13-step process done jointly by MELP and MOF specialists (Resources Inventory Committee 1998*b*). Habitat assessment ratings of capability and suitability values will also have a similar, formalized multistep process conducted by MELP Victoria and regional experts (Resources Inventory Committee 1998*c*). For inventories of wildlife species, including identified wildlife and other significant species, quality assurance will be delivered through spot checks by MELP species' experts (Resources Inventory Committee 1998*d*).

The first step in quality assurance is done by regional MELP Wildlife and Habitat Inventory Specialists (WHISs). For many inventories (e.g., bats) quality assurance will be done centrally in Victoria, at the discretion of the WHISs.

Database data entry and storage standards are being developed for species and habitat inventory. These will include standards control, contract monitoring, and guidance.

LINKAGES TO OTHER INVENTORIES

Through the Resources Inventory Committee the Province of British Columbia has defined a hierarchical ecosystem classification that has 4 levels: regional, local, vegetation development, and field data collection (Fig. 2). Each of those levels can be further subdivided. For example, the regional ecosystem level is composed of the ecoregion and the zonal classifications, and each of those classifications can be further subdivided into classes: 5 for the ecoregions and 4 for the zonal classification.

Wildlife species and wildlife habitat inventories are often

directly or indirectly linked to other resource inventories. This can mean in some cases supplying required data or being an interim step in a more detailed inventory. In other cases, such as the highest level of habitat inventory, ecoregion mapping is partly based on a product (biogeoclimatic ecosystem classification) that has been developed and mapped by the MOF. So, even though the 2 classifications are mutually exclusive, they support each other.

Vegetation Resources Inventory (VRI) is enhanced forest cover mapping, whereby understory vegetation, structural features, and soils are used to define the mapped units (Resources Inventory Committee 1997, 1998*e*). The VRI is designed to gather information on tree species, height, age, and productivity, which is vital information in determining structural stages for TEM. The VRI is also used to gather physiognomic information about non-treed areas, such as grasslands, wetlands, alpine, and recently clearcut. Such information is useful in determining general land classes for Broad Ecosystem Inventory (BEI), and as a surrogate product to TEM for assigning habitat values to identified wildlife.

Terrain mapping is a requirement for determining slope stability, road location, and terrain hazards. This mapping can be used as the first step in TEM and VRI to delineate landscape polygons for assigning site series or vegetation classes.

Species and habitat inventory are also linked; for example, habitat inventory is a stratification tool for species' surveys, such as stratified random sampling. Also, habitat inventory is used to support species population estimates. Species inventories are essential in providing quantified data for fine tuning capability and suitability habitat assessment ratings.

Land and Resource Management Planning (LRMP) and Commission on Resources and Environment (CORE) areas are higher level plans that will be developed for 22 areas, covering most of the province. They form a bridge between national and provincial strategic resource use plans and landscape unit, forest development, and resource use plans. Wildlife species and wildlife habitat inventories for LRMP and CORE areas are at a broad level, but provide guidance for prioritizing areas requiring more detailed inventories.

Native land claim and other resource allocation issues often require spatial as well as tabular information about the distribution, abundance, and status of wildlife species. Such information can be gathered at a broad or general inventory

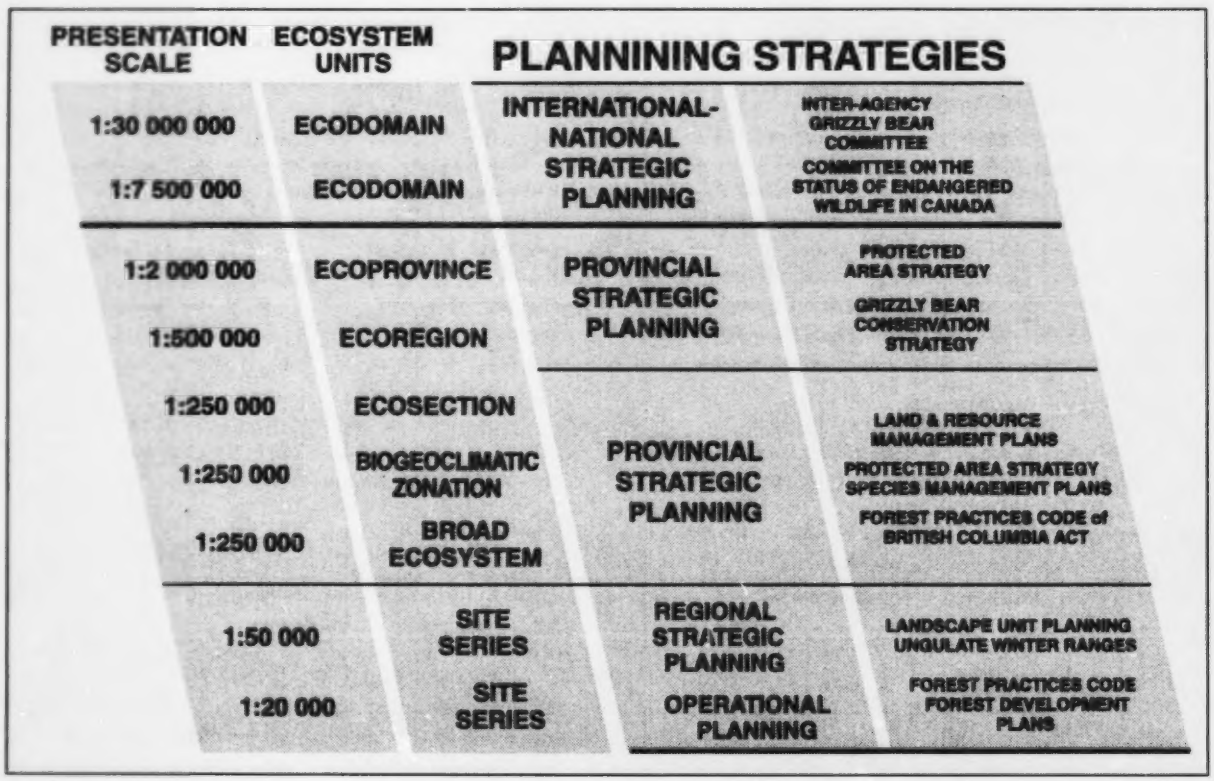

Figure 2. Ecosystem identification level and map scale used in various land-based planning strategies that have been initiated in British Columbia.

level, but specific issues for protection, conservation, or resource sharing may require detailed (1:20,000 TEM and absolute abundance inventory of a species) inventories for transferring resource or management responsibilities.

ACKNOWLEDGEMENTS

We thank Diana Demarchi for preparing the figures.

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Gap Analysis for Ecosystems and Species at Risk Using Terrestrial Ecosystem Mapping

Carmen Cadrin
Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8T 9M1, Canada
Carmen.Cadrin@gems3.env.gov.bc.ca

Barb von Sacken
Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8T 9M1, Canada

ABSTRACT

Terrestrial Ecosystem Mapping (TEM) is the standard methodology used in British Columbia to map terrestrial ecosystems. TEM uses a 3-level classification hierarchy: ecoregion units and biogeoclimatic units at broader levels, and site units with vegetation development stages at the detailed level. Accompanying databases consist of a core set of data attributes, which can be used to develop interpretive products for a variety of purposes, including management interpretations for species and ecosystems at risk. This paper provides a summary of TEM projects completed or ongoing in British Columbia as of August 1998. TEM coverage was analyzed, and its usefulness to the management of species and ecosystems at risk was assessed, using both the ecoregion and biogeoclimatic classifications and the British Columbia Conservation Data Centre (CDC) rare element occurrence database. It was determined that the majority of TEM completed to date did not include areas of high rarity value. Recommendations for future TEM in the province based on distribution of ecosystems and species at risk are discussed.

Key words: biogeoclimatic zone, ecosection, ecoprovince, gap analysis, mapping methodology, terrestrial ecosystem mapping.

The Standard for Terrestrial Ecosystem Mapping (TEM; RIC 1998) is a methodology designed for mapping at scales of from 1:5,000 to 1:50,000. It builds upon existing methodologies and classifications (Fig. 1) developed by the British Columbia Ministry of Forests (MOF; Mitchell et al. 1989) and the British Columbia Ministry of Environment, Lands and Parks (MELP; Demarchi et al. 1990). The flexibility of TEM allows for a wide variety of interpretations, which provide management information for species and habitats at risk. Since TEM's introduction in 1995, nearly 100 TEM projects have been initiated provincially.

The objective of this paper is to initiate a strategic plan for future TEM in British Columbia via a gap analysis of current TEM. Existing TEM is compared to distribution of rare species (birds, animals, and plants) and rare ecosystems, as defined by the British Columbia Conservation Data Centre (CDC)[1] (B.C. CDC 1999). This analysis uses 2 levels of classification in the TEM methodology: ecoregion classification (Demarchi 1996), and biogeoclimatic classification (Meidinger and Pojar 1991).

[1] For a description of the Conservation Data Centre, visit their home page at http://www.env.gov.bc.ca/wld/cdc.

TEM STANDARDS AND INTERPRETATIONS

TEM guidelines, combined with a core set of data attributes required for each TEM project, regardless of project objectives, provide a powerful provincial planning tool. By having a standard dataset, data warehouse, and format for data presentation, information can be readily accessed and shared by a wide variety of users. In the common inventory and mapping language that TEM provides, we can further build on our provincial knowledge base to develop a better understanding

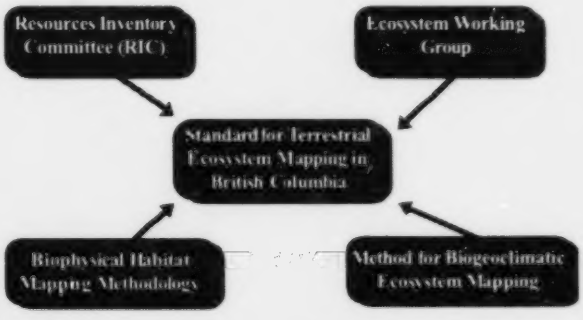
Resources Inventory Committee (RIC)
Ecosystem Working Group
Standard for Terrestrial Ecosystem Mapping in British Columbia
Biophysical Habitat Mapping Methodology
Method for Biogeoclimatic Ecosystem Mapping

Figure 1. Development of Terrestrial Ecosystem Mapping Standards.

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of the scope and breadth of ecological diversity that exists provincially. TEM data can be used across jurisdictions, regions, client groups, and projects. The provincial data warehouse will serve to provide baseline information for future management and planning strategies. The analysis presented in this paper illustrates the ability we now have to assess gaps in inventory and mapping of provincially rare species habitat and ecosystems.

Some interpretive products from TEM focus specifically on species or ecosystems at risk (e.g., grizzly bear or marten habitat capability and suitability), while others focus on resource management (e.g., forest harvesting, silviculture treatments, urban planning exercises, conservation of natural areas, or planning of pipeline corridors). While many management practices are not aimed specifically at species at risk, they affect the plant and animal species that utilize the ecosystem as habitat, and the ecosystem itself. Regardless of the form of resource management, whether it be for mining, urban development, or forest management, there are implications for the biology and management of species and ecosystems at risk.

STATUS OF TEM

Along with the first draft of TEM in 1995, a significant number of TEM projects at scales of 1:50,000 and 1:20,000 were also initiated (Fig. 2). Of the approximately 95 million hectares of land in British Columbia, approximately 9.3% has been mapped using TEM. Approximately 5% has been mapped at 1:50,000, and 4.3% has been mapped at 1:20,000. The number of TEM projects has continued to grow, totalling 99 projects province-wide as of 31 August 1998. These initial TEM projects have depended largely on the interests of specific clients (Fig. 3) and regional access to Forest Renewal British Columbia (FRBC) funding.

While initial TEM projects may have addressed species or ecosystems at risk, initiation of these projects was not based on a provincial mapping strategy or priority list. As funding sources and the understanding of the potential interpretative uses of TEM evolved, so have client groups of TEM. For example, the Columbia Basin Fish and Wildlife Compensation Program in the Kootenays has completed a significant amount of TEM in order to assess wildlife values within its management areas. Increasingly, TEM projects are being initiated by local governments, First Nations, the British Columbia Ministry of Energy and Mines, and British Columbia Parks, illustrating a shift in applications of TEM.

METHODS

The analysis of TEM coverage in British Columbia demonstrates that almost 10% of the province is mapped at 1:20,000 or 1:50,000. A detailed TEM project tracking system maintained by MELP (B.C. MELP 1998) includes a map of all TEM projects currently underway (Fig. 2). Biogeoclimatic maps and ecoregion maps were used in conjunction with the TEM base map to arrive at area summaries for all biogeoclimatic and ecoregion units with TEM coverage.[2] It is important to note that other mapping methodologies, such as Biophysical Habitat Mapping (Demarchi et al. 1990) and Sensitive

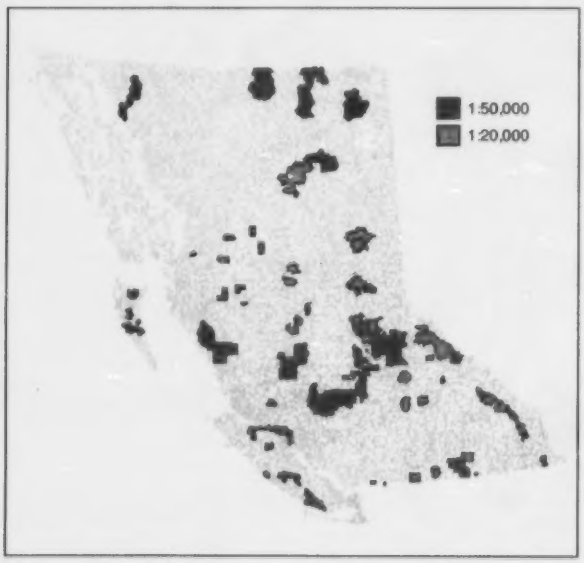

Figure 2. Distribution of TEM mapping in British Columbia.

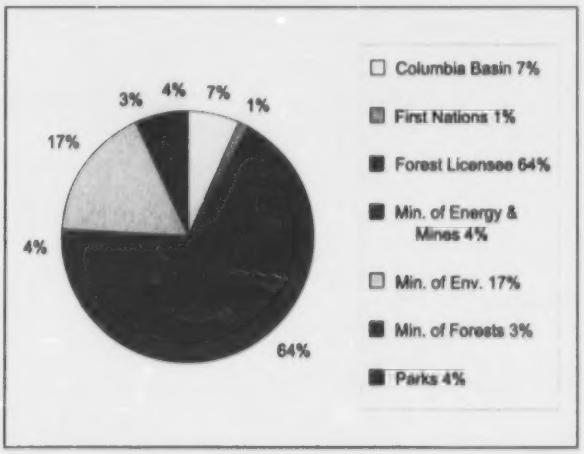

Figure 3. TEM projects by client type.

[2] Area summaries are approximate, reflecting differences in map scale and generalizations in TEM project boundaries.

Ecosystem Inventory (Ward et al. 1999), provide inventory and mapping for significant portions of ecoregions and biogeoclimatic units. For the purposes of this analysis these other methodologies are not considered. In some cases, such as the Southern Interior ecoprovince, the existence of other mapping may have played a factor in the lack of TEM, and/or could be revised to TEM standards and incorporated into the provincial coverage of TEM. TEM coverage was compared to areas of high rarity value to determine how effective the current TEM is in providing tools to develop management plans for species and ecosystems at risk.

The analysis uses high rarity value to rank ecoregion and biogeoclimatic units for mapping priorities. The criteria for determining areas of high rarity value are based on the CDC rare element database and tracking lists (B.C. CDC 1999). A rare element occurrence (EOR) is the actual location of a threatened or endangered plant, animal, or plant association[3] as identified by the CDC. An ecosection with high rarity value is one with >75 EORs or more than one-third of its ecosystems listed as rare by the CDC. A biogeoclimatic unit with high rarity value is one with >50 EORs or more than one-third of its ecosystems listed as rare by the CDC tracking list. A comparison of the area of TEM within each identified ecoregion and biogeoclimatic unit indicated gaps in inventory data relevant to the occurrence of rare elements in British Columbia. The number of EORs and rare ecosystems within the ecoregion and biogeoclimatic units, in conjunction with TEM coverage, determined the ranking priority for future TEM.

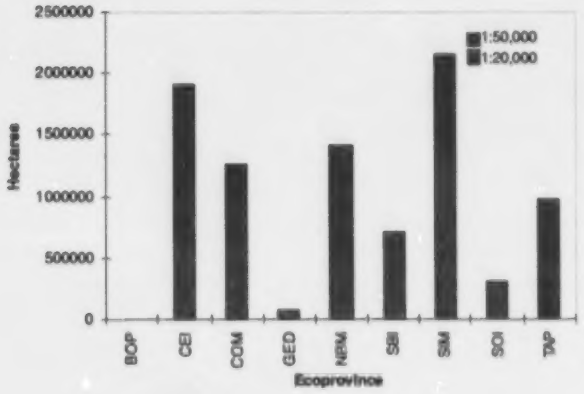

Figure 4. TEM mapping by ecoprovince (see Table 1 for codes).

[3] The ecosystem is a member of a specific plant association and, for this paper, the term "ecosystem" is treated as equivalent to the rare plant association.

RESULTS

TEM Gap Analysis at the Ecoregion Level

TEM has been conducted in all 9 of the terrestrial ecoprovinces, although in the Boreal Plains ecoprovince there is so little TEM coverage that it comprises <0.5% (Fig. 4). The majority of the mapping, 95% (Fig. 5), occurs in 6 ecoprovinces: Southern Interior Mountains; Central Interior; Northern Boreal Mountains; Coast and Mountains; Taiga Plains; and Sub-Boreal Interior. Only 5% of the mapping occurs in the remaining 3 ecoprovinces: Georgia Depression; Southern Interior; and Boreal Plains. These 3 are the smallest ecoprovinces (Table 1), and while one would expect areas of mapping to reflect that size difference, the contrast in total area mapped between the 2 groups is significant.

The analysis highlights 18 ecosections that fall within the criteria of high rarity value (Table 2). The results are even more significant than expected: 12 of the 18 ecosections occur within the 3 ecoprovinces with the least amount of TEM (Georgia Depression, Southern Interior, and Boreal Plains). Eleven of these ecosections occur within the Georgia Depression and the Southern Interior. The ecoprovinces with the highest amount of TEM (Southern Interior Mountains and Central Interior), each include only 1 ecosection with high rarity value.

The Coast and Mountains is a wide-ranging ecoprovince, which includes the Queen Charlotte Islands, and most of Vancouver Island and the coastal mainland. Of the 25 ecosections within this ecoprovince, 4 (Eastern Pacific Ranges, Windward Island Mountains, Skidegate Plateau, and Windward Queen Charlotte Mountains) are rated high in rarity value. These ecosections range from having almost no mapping to moderate amounts of TEM (Table 2).

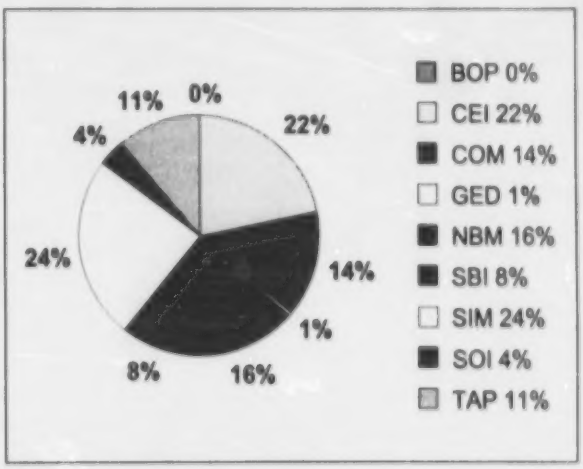

Figure 5. Total distribution of TEM by ecoprovince (see Table 1 for codes).

Table 1. Percentage of TEM coverage by terrestrial ecoprovince.

Ecoprovince	Code	Total area (ha)	% mapped
Coast and Mountains	COM	18,841,854	3
Georgia Depression	GED	1,833,924	4
Central Interior	CEI	10,489,177	6
Sub-Boreal Interior	SBI	14,200,964	5
Southern Interior Mountains	SIM	13,675,827	9
Southern Interior	SOI	5,832,390	5
Boreal Plains	BOP	3,952,454	0
Taiga Plains	TAP	5,971,567	14
Northern Boreal Mountains	NBM	18,982,157	3

The Georgia Depression includes 7 ecosections, 6 of which are rated high for rarity values (Fig. 6). The seventh ecosection, Juan de Fuca Strait, is a marine ecosection. The Georgia Lowland ecosection is low in rare element occurrences, but high in ecosystem rarity and, as such, is particularly vulnerable to increasing population pressures.

The Southern Interior ecoprovince is another area high in rarity value. Of the 12 ecosections, 5 (Southern Thompson Upland, Thompson Basin, Northern Okanagan Highland, Okanagan Range, and Southern Okanogan Basin) are rated high for the numbers of rare element occurrences and rare ecosystems. All except the Southern Okanogan Basin have little or no TEM coverage.

The Boreal Plains is a smaller ecoprovince located in the northeast portion of the province and is an extension of the much more wide-ranging central plains of Canada. One of its 4 ecosections, the Peace Lowland, has high rarities in the CDC element database and rare grassland ecosystems.

Of the 12 ecosections within the Central Interior ecoprovince, the Fraser River Basin is the only one rated high for rarity value. This ecosection has TEM coverage of 65% of the area.

The Southern Interior Mountains ecoprovince includes 19 ecosections, 1 of which was rated high for rarity value. The East Kootenay Trench has moderate TEM coverage at 24% of the area.

The Northern Boreal Mountains, the Sub-Boreal Interior, and the Taiga Plains included no ecosections rated high for rarity value.

TEM Gap Analysis at the Biogeoclimatic Level

All of the 14 biogeoclimatic zones within British Columbia have some degree of TEM (Fig. 7). The distribution of total TEM by zone is shown in Figure 8. A total of 71% of TEM

Table 2. Priority ranking for ecosections rated as having high rarity values. (EOR = rare element occurrence.)

Ecosection	Code	No. of EORs	No. of rare ecosystems	% mapped	Priority ranking
Eastern Pacific Ranges	EPR	101	18	0	1
Windward Queen Charlotte Mountains	WQC	192	22	6	1
Fraser Lowland	FRL	197	42	1	1
Leeward Island Mountains	LIM	174	27	6	1
Nanaimo Lowland	NAL	638	56	0	1
Southern Gulf Islands	SGI	193	43	0	1
Strait Of Georgia	SOG	135	43	0	1
Northern Okanagan Highland	NOH	118	14	6	1
Southern Thompson Upland	STU	124	32	0	1
Thompson Basin	THB	86	16	4	1
Georgia Lowland	GEL	29	40	0	2
Peace Lowland	PEL	101	2	0	2
Skidegate Plateau	SKP	134	10	17	3
Windward Island Mountains	WIM	327	23	11	3
East Kootenay Trench	EKT	147	10	24	3
Okanagan Range	OKR	178	26	25	3
Fraser River Basin	FRB	115	22	65	4
Southern Okanogan Basin	SOB	364	25	14	4

Table 3. Percentage of TEM coverage by biogeoclimatic (BGC) zone.

BGC zone	Total area (ha)	% mapped	BGC zone	Total area (ha)	% mapped
Alpine Tundra and Parkland	17,757,600	7	Interior Douglas-fir	4,372,100	13
Bunchgrass	319,100	16	Mountain Hemlock	4,027,500	3
Boreal White and Black Spruce	14,831,400	10	Montane Spruce	2,642,900	18
Coastal Douglas-fir	205,900	0	Ponderosa Pine	340,400	4
Coastal Western Hemlock	10,607,600	9	Sub-Boreal Pine–Spruce	2,374,100	9
Engelmann Spruce–Subalpine Fir	13,156,100	12	Sub-Boreal Spruce	10,080,700	9
Interior Cedar–Hemlock	5,238,800	11	Spruce–Willow–Birch	8,270,300	5

occurs in the following 5 zones: Alpine Tundra and Parkland (AT&P)[4]; Boreal White and Black Spruce (BWBS); Coastal Western Hemlock (CWH); Engelmann Spruce–Subalpine Fir (ESSF); and Sub-Boreal Spruce (SBS). However, the distribution of mapping relative to total area of a biogeoclimatic zone varies widely (Table 3). Zones that have a very small total area can have a significant amount of TEM. For example, TEM covers 16% of the Bunchgrass (BG) zone's total area, while there is <0.01% TEM coverage in the Coastal Douglas-fir (CDF) zone. Zones covering large areas of the province also have variable TEM coverage. For example, 12% of the ESSF has TEM coverage, compared to 7% of the AT&P.

Assessing TEM coverage by biogeoclimatic zone provides an overview of where mapping occurs in the province. A finer level of detail, the biogeoclimatic subzone/variant level, is required to assess gaps in mapping for rare element occurrences (EORs) and rare ecosystems.

There are 133 biogeoclimatic subzone/variants currently described within British Columbia. Analysis of TEM by subzone/variant identifies 46 that meet the criteria for high rarity (Table 4). The distribution of the 46 units shows 4 zones as having the most significant number of EORs and rare ecosystems (Table 5). The BG, CWH, Ponderosa Pine (PP), and Interior Douglas-fir (IDF) zones have the highest percentages of units with high numbers of rarities. The CDF has only 1 subzone unit described; it is rated high in rarity values.

The Alpine Tundra and Parkland zone is not further discussed in this document. This zone is widely distributed at high elevations throughout British Columbia, but its plant communities are poorly described and correlated, making it difficult to determine gaps in TEM and their relation to rare species and ecosystems.

The BG, located in the dry southern Interior of the province, has 5 biogeoclimatic units, all of which are considered to have high rarity values. While the BG is limited in its distribution, 16% of the total area has been mapped. However, the bulk of the mapping has focused on the BGxw2 and BGxh3 subzone/variants (Table 6). While both of these biogeoclimatic units have significant numbers of EORs and rare ecosystems, the remaining 3 units (BGxh1, BGxh2, and BGxw1), have very little to no TEM and the highest numbers of rarities. Biophysical habitat mapping in the South Okanagan (Lea et al. 1991) focused on the BGxh1 subzone and could be updated and incorporated into the scope of TEM coverage.

The CDF is limited to the southeast portions of Vancouver Island, the Gulf Islands, and the Sunshine Coast. Much of the

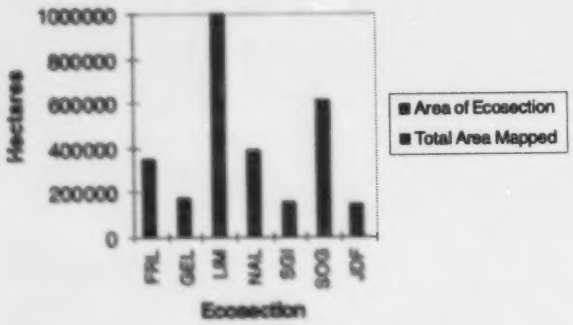

Figure 6. TEM by ecosections of the Georgia Depression ecoprovince (see Table 2 for codes).

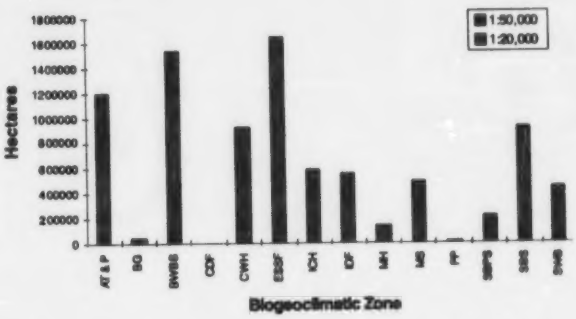

Figure 7. TEM by biogeoclimatic zone (see Table 5 for codes).

[4] At regional and provincial mapping scales, biogeoclimatic mapping combines parkland with the Alpine Tundra (AT) zone, rather than mapping the parkland with the adjacent forested subzone. This paper follows this convention for area calculations.

Table 4. Codes and names of biogeoclimatic (BGC) units (subzones/variants; B.C. MOF and B.C. MELP 1998) that meet the criteria for high rarity.

BGC unit code	BGC unit name
BGxh1	Okanagan - Very Dry Hot - Bunchgrass
BGxh2	Thompson - Very Dry Hot - Bunchgrass
BGxh3	Fraser - Very Dry Hot - Bunchgrass
BGxw1	Nicola - Very Dry Warm - Bunchgrass
BGxw2	Alkali - Very Dry Warm - Bunchgrass
BWBSmw1	Peace - Moist Warm - Boreal White and Black Spruce
CDFmm	Moist Maritime - Coastal Douglas-fir
CWHdm	Dry Maritime - Coastal Western Hemlock
CWHds1	Central - Dry Submaritime - Coastal Western Hemlock
CWHds2	Southern - Dry Submaritime - Coastal Western Hemlock
CWHmm1	Submontane - Moist Maritime -Coastal Western Hemlock
CWHmm2	Montane - Moist Maritime - Coastal Western Hemlock
CWHms1	Southern - Moist Submaritime - Coastal Western Hemlock
CWHms2	Central - Moist Submaritime - Coastal Western Hemlock
CWHvh1	Southern - Very Wet Hypermaritime - Coastal Western Hemlock
CWHvh2	Central - Very Wet Hypermaritime - Coastal Western Hemlock
CWHvm1	Submontane - Very Wet Maritime - Coastal Western Hemlock
CWHvm2	Montane - Very Wet Maritime - Coastal Western Hemlock
CWHwh1	Submontane - Wet Hypermaritime - Coastal Western Hemlock
CWHwm	Wet Maritime - Coastal Western Hemlock
CWHws1	Submontane - Wet Submaritime - Coastal Western Hemlock
CWHws2	Montane - Wet Submaritime - Coastal Western Hemlock
CWHxm1	Eastern - Very Dry Maritime - Coastal Western Hemlock
CWHxm2	Western - Very Dry Maritime - Coastal Western Hemlock
ESSFdk	Dry Cool - Engelmann Spruce–Subalpine Fir
ICHdw	Dry Warm - Interior Cedar–Hemlock
ICHmk3	Horsefly - Moist Cool - Interior Cedar–Hemlock
ICHmm	Moist Mild - Interior Cedar–Hemlock
IDFdk1	Thompson - Dry Cool - Interior Douglas-fir
IDFdk3	Fraser - Dry Cool - Interior Douglas-fir
IDFdk4	Chilcotin - Dry Cool - Interior Douglas-fir
IDFdm1	Kettle - Dry Mild - Interior Douglas-fir
IDFdm2	Kootenay - Dry Mild - Interior Douglas-fir
IDFww	Wet Warm - Interior Douglas-fir
IDFxh1	Okanagan - Very Dry Hot - Interior Douglas-fir
IDFxh2	Thompson - Very Dry Hot - Interior Douglas-fir
IDFxm	Very Dry Warm - Interior Douglas-fir
IDFxw	Very Dry Warm - Interior Douglas-fir
MHmm1	Windward - Moist Maritime - Mountain Hemlock
PPdh1	Kettle - Dry Hot - Ponderosa Pine
PPdh2	Kootenay - Dry Hot - Ponderosa Pine
PPxh1	Okanagan - Very Dry Hot - Ponderosa Pine
PPxh2	Thompson Very Dry Hot - Ponderosa Pine
SBSdh	Dry Hot - Sub-Boreal Spruce
SBSdk	Dry Cool - Sub-Boreal Spruce
SBSmh	Moist Hot - Sub-Boreal Spruce

Table 5. Number of biogeoclimatic (BGC) units (subzones/variants; B.C. MOF and B.C. MELP 1998) with high rarity values.

BGC zone	BGC code	Total no. of units	No. of units with high rarity values	% of units with high rarity values
Alpine Tundra and Parkland	AT&P	N/A	N/A	N/A
Bunchgrass	BG	5	5	100
Coastal Douglas-fir	CDF	1	1	100
Coastal Western Hemlock	CWH	19	17	89
Ponderosa Pine	PP	4	4	100
Interior Douglas-fir	IDF	15	10	67
Sub-Boreal Spruce	SBS	17	3	18
Interior Cedar–Hemlock	ICH	20	3	15
Boreal White and Black Spruce	BWBS	8	1	12
Montane Spruce	MS	8	0	0
Engelmann Spruce–Subalpine Fir	ESSF	25	1	.04
Mountain Hemlock	MH	4	1	25
Sub-Boreal Pine–Spruce	SBPS	4	0	0
Spruce–Willow–Birch	SWB	3	0	0
Total		133	46	

CDF has been altered through urban and resource development. The remaining natural areas play a critical role in providing habitat for species and representation of ecosystems at risk. There are only a few very small TEM projects completed in the CDFmm (<0.01% of the zone).

The wide-ranging Coastal Western Hemlock zone stretches along the coast and coast mountains from the Canada–United States border to northern British Columbia. This zone has 19 biogeoclimatic units described, of which 17 are considered to have high rarity values (Table 6). The amount of TEM in each varies significantly, from 0% in the CWHds2 to >20% in the CWHmm1. This zone is the single most significant zone in terms of mapping recommendations, even though 9% of its total area has been mapped.

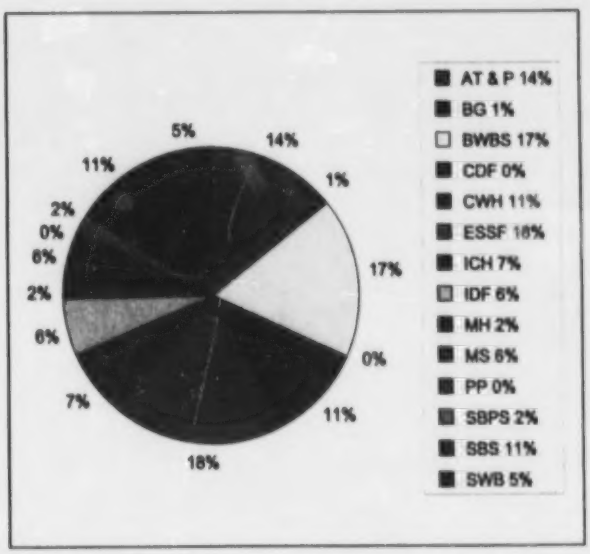

Figure 8. Total distribution of TEM by biogeoclimatic zone (see Table 5 for codes).

The Ponderosa Pine zone is the smallest biogeoclimatic zone. Its distribution is limited to the lower valleys of the Fraser, Thompson, Okanagan, and Nicola rivers, and the southern portion of the Rocky Mountain Trench. Only 4% of its total area has been mapped. There are 4 biogeoclimatic units in the zone, all of which are considered high priority (Table 6). One subzone, the PPxh1, has also been mapped in the South Okanagan (Lea et al. 1991).

The Interior Douglas-fir zone, occurring throughout the southern Interior and East Kootenays, has 15 biogeoclimatic units. Of these, 10 have high rarity values. While 13% of the IDF's total area has been mapped, mapping within areas of high priority varies significantly (Table 6).

The remaining 9 biogeoclimatic units with high rarity values occur within the Sub-Boreal Spruce, the Interior Cedar–Hemlock (ICH), the Boreal White and Black Spruce, the Engelmann Spruce–Subalpine Fir, and the Mountain Hemlock (MH) zones (Table 6). The Sub-Boreal Pine–Spruce (SBPS), the Montane Spruce (MS), and the Spruce–Willow–Birch (SWB) zones have no units with high rarities.

DISCUSSION

An inventory strategy for the protection of rare wildlife, plants, and ecosystems in British Columbia can be developed from this TEM Gap Analysis. Recommendations for future TEM are based on the amount of existing TEM coverage and the rarity values of each biogeoclimatic or ecoregion unit.

Determining Priorities for Ecoregion Units

The analysis identifies 18 ecosections with high numbers of EORs or rare ecosystems. Of these 18 ecosections, 6 have significant (>10%) amounts of TEM. These ecosections are assigned lower priority than those ecosections with <10% TEM. Table 2 ranks the 18 ecoregions as Priority 1–4 according to the following criteria:

- Priority 1: high numbers of EORs and rare ecosystems, and <10% TEM coverage.
- Priority 2: high numbers of EORs or high numbers of rare ecosystems, <10% TEM coverage, and >10 rare ecosystems.
- Priority 3: high numbers of EORs and/or high numbers of rare ecosystems, and >10% TEM coverage.
- Priority 4: high numbers of EORs and/or high numbers of rare ecosystems, and >50% TEM coverage. It also includes the areas with mapping easily converted to TEM.

TEM is highly recommended for 2 ecosections within the Coast and Mountains ecoprovince. The Eastern Pacific Ranges ecosection on the mainland occurs in the area of transition between the coastal and interior climates, resulting in ecosystems particularly high in species diversity. For example, dry Ponderosa Pine forest and moist Cedar-Hemlock forest are located within a few kilometres of each other within the Skagit Valley. The Windward Queen Charlotte Mountains ecosection is also high in priority. Many endemic plant species, known only from glacial refugia, occur in this area. Two other ecosections are recommended for future TEM but are third in priority. These are the Skidegate Plateau, also on the Queen Charlotte Islands, and the Windward Island Mountains on the west side of Vancouver Island, an area known for its coastal temperate rain forests.

The 6 terrestrial ecosections of the Georgia Depression are recommended for future TEM (Table 2). In addition to the natural causes of rarity in this ecoprovince, intense population and development pressures have resulted in extensive depletion of the native species and ecosystems that once occurred more commonly throughout the area. In some selected areas, local governments and British Columbia Parks are initiating TEM at larger scales (1:10,000 and 1:5,000) for planning and conservation management. It is important to note that the Sensitive Ecosystem Inventory (Ward et al. 1999) provides partial mapping of the Nanaimo Lowland, the Southern Gulf Islands, and a portion of the Strait of Georgia, identifying generalized rare ecosystem units. Such partnerships amongst agencies, both governmental and non-governmental, are recommended to complete the work in this biologically diverse area.

Within the Southern Interior ecoprovince, 4 ecosections are recommended for further TEM. This area, also high in natural forms of rarity, has experienced loss of species and ecosystems in the past from agricultural and forestry land

Table 6. Priority ranking for biogeoclimatic (BGC) units rated as having high rarity values. See Table 4 for codes. (EOR = rare element occurrence.)

BGC unit	No. of EORs	No. of rare ecosystems	% mapped	Priority ranking
BGxh1	315	11	0.1	1
BGxh2	42	6	0.2	3
BGxh3	23	2	66.9	4
BGxw1	40	5	0.0	3
BGxw2	57	3	52.7	4
BWBSmw1	156	0	1.6	2
CDFmm	759	18	0.0	1
CWHdm	144	13	3.7	1
CWHds1	30	9	0.1	3
CWHds2	2	10	0.0	3
CWHmm1	6	7	20.3	4
CWHmm2	12	6	3.9	3
CWHms1	46	7	6.0	3
CWHms2	2	7	0.0	3
CWHvh1	280	12	2.1	1
CWHvh2	150	11	13.4	1
CWHvm1	132	7	4.8	1
CWHvm2	35	4	2.7	3
CWHwh1	92	8	14.5	1
CWHwm	4	7	10.1	3
CWHws1	3	5	0.5	3
CWHws2	0	6	2.6	3
CWHxm1	177	16	1.7	1
CWHxm2	104	14	10.8	1
ESSFdk	54	0	7.5	2
ICHdw	62	0	11.0	2
ICHmk3	0	4	6.7	3
ICHmm	0	3	43.4	4
IDFdk1	35	4	0.9	3
IDFdk3	36	7	21.3	4
IDFdk4	13	8	3.6	3
IDFdm1	21	3	12.8	3
IDFdm2	71	5	15.1	1
IDFww	14	4	0.1	3
IDFxh1	159	10	4.8	1
IDFxh2	38	7	2.5	3
IDFxm	55	6	58.6	4
IDFxw	4	6	0.0	3
MHmm1	53	0	3.7	2
PPdh1	25	5	12.4	3
PPdh2	38	5	3.3	3
PPxh1	118	9	4.1	1
PPxh2	24	6	3.4	3
SBSdh	7	4	39.1	4
SBSdk	16	4	2.5	3
SBSmh	7	7	1.4	3

Table 7. Overlap of ecosections and biogeoclimatic (BGC) units with high rarity values. See Tables 2 and 4 for codes.

Ecosection	BGC unit(s)
EPR	CWHds1, CWHms1, IDFww
SKP	CWHvh2, CWHwh1
WQC	CWHvh2, CWHwh1
WIM	CWHmm1, CWHvh1, CWHvm1, CWHvm2, CWHxm2, MHmm1
GEL	CWHdm, CWHmm1, CWHvm1, CWHvm2, CWHxm1
NAL	CDFmm, CWHmm1, CWHmm2, CWHxm1, CWHxm2
FRL	CDFmm, CWHdm, CWHds1, CWHvm2, CWHxm1
SGI	CDFmm, CWHxm1, CWHxm2
SOG	CDFmm, CWHmm1, CWHxm1, CWHxm2,
LIM	CWHmm1, CWHmm2, CWHvm2, CWHxm1, CWHxm2, MHmm1
FRB	BGxh1, BGxh3, BGxw2, IDFdk4, IDFxm
EKT	ESSFdk, IDFdm2, PPdh1, PPdh2
SOB	BGxh1, IDFxh1, PPxh1
NOH	IDFdm1, IDFxh1, PPdh1, PPdh2
STU	BGxh2, IDFdk1, IDFdm2, IDFxh2, PPxh2
THB	BGxh2, BGxw1, IDFdk1, IDFxh1, IDFxh2, IDFxw, PPxh2
OKR	BGxh1, IDFdk1, IDFdm2, IDFxh1, PPxh1
PEL	BWBSmw1

use, and now from increasing population pressures. The Southern Thompson Upland, the Thompson Basin, and the Northern Okanagan Highland should be the first priority for future TEM. The Okanagan Ranges and the South Okanogan Basin are third and fourth, respectively, in priority, as each has >10% TEM coverage. There has been considerable detailed mapping using the Biophysical Inventory Methodology (Demarchi et al. 1990) for the South Okanogan Basin ecosection (Lea et al. 1991). With some adaptation of attribute codes, this project could be used in the TEM coverage. Therefore, the priority for the Okanogan Basin ecosection, rather than mapping, would be to convert the existing biophysical mapping to the standard TEM methodology for inclusion in the provincial data warehouse.

The Peace Lowland, within the Boreal Plains ecoprovince, is ranked high in rarity value because of its high number of EORs. It is ranked priority 2 for future TEM. The East Kootenay Trench includes both high numbers of EORs and rare ecosystems, but because of its relatively high amount of TEM, is ranked third in mapping priority. The Fraser River Basin, while high in both EORs and rare ecosystems, has TEM coverage for more than half of its area. Therefore it is ranked fourth in mapping priority and is not recommended for future mapping.

Determining Priorities for Biogeoclimatic Units

The analysis identifies 46 biogeoclimatic units as high in EORs and/or rare ecosystems. Of these 46 units, 7 have significant amounts of TEM (i.e., >20% of the total subzone mapped). Those subzones that are considered to have significant amounts of TEM are assigned lower priority than those with very little TEM. The remaining subzones have the following breakdown of rarities: 12 have high numbers of both EORs and rare ecosystems; 4 have high numbers of EORs only; and 23 have high numbers of rare ecosystems only. Table 6 ranks the 46 subzones as Priority 1–4 according to the following criteria:

- Priority 1: high numbers of EORs and rare ecosystems, and <20% TEM coverage.
- Priority 2: high numbers of EORs only, and <20% TEM coverage.
- Priority 3: high numbers of rare ecosystems only, and <20% TEM coverage.
- Priority 4: high numbers of EORs and/or rare ecosystems, and >20% TEM coverage.

The 5 zones with the most significant numbers of units with high rarities and limited mapping are the CWH, CDF, PP, IDF, and BG.

SUMMARY

The analysis indicates a high degree of overlap between the 18 ecosections in British Columbia that have been identified as having high rarity values and the 46 biogeoclimatic units with high rarity values (Table 7). Each of the listed ecosections includes at least 1 biogeoclimatic unit that is high either in rare element occurrences or in rare ecosystems. We recommend 16 of these ecosections for future Terrestrial Ecosystem Mapping with wildlife and rare ecosystem interpretations. The Fraser River Basin and the South Okanogan Basin ecosections are not recommended for further mapping at this time. The biogeoclimatic units that occur within these ecosections (BGxh3, BGxw2, IDFdk4, and IDFxm), have high TEM coverage or are mapped (Lea et al. 1991), and are therefore not considered in the following discussion of biogeoclimatic units.

Of the 133 biogeoclimatic units in British Columbia, 46 are rated high in rarity values. The overlap of ecosection and biogeoclimatic units indicates that 34 of the biogeoclimatic units will be represented by mapping within the high priority ecosections.

Twelve biogeoclimatic units are not represented in the ecosection overlap. Of these 12 biogeoclimatic units, the ICHmm, IDFdk3, and SBSdh are ranked fourth in priority because of significant existing TEM coverage, and are therefore not further recommended for mapping. This leaves 9 biogeoclimatic units recommended for further mapping. They are: CWHds2, CWHms2, CWHwm, CWHws1, CWHws2,

ICHdw, ICHmk3, SBSdk, and SBSmh.

Terrestrial Ecosystem Mapping of project areas within the 16 above-listed ecosections and 9 biogeoclimatic units will provide considerable inventory data for a significant proportion of the species and ecosystems at risk in British Columbia. Information on the location of rare species' habitats and rare ecosystems can be used to prepare management plans that will, in future, protect the rare elements of biodiversity in British Columbia.

ACKNOWLEDGEMENTS

We thank D. Webb for her invaluable assistance in preparing the figures and tables, and in the final formatting of the paper. We also thank B. Low for preparing the summary statistics of Terrestrial Ecosystem Mapping in British Columbia.

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Deriving Wildlife Habitat Values From Ecosystem Maps

Lynne Bonner
Wildlife Inventory Section, Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8T 9M1, Canada
Lynne.Bonner@gems1.gov.bc.ca

Dennis A. Demarchi
Wildlife Inventory Section, Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8T 9M1, Canada

ABSTRACT

A provincial standard for assessing wildlife habitat has been developed for British Columbia. This paper outlines the standard definitions and habitat rating criteria used in BC Wildlife Habitat Rating Standards. Wildlife habitat capability and suitability ratings can be applied to ecological mapping at scales from 1:250,000 to 1:20,000. Ratings indicate the value of a habitat to support a particular wildlife species for a specified habitat use compared to the best habitat in the province (the provincial benchmark). Rating criteria are defined for different map scales and different levels of detail. A wildlife habitat capability and suitability assessment project requires development of species-habitat models that are ground-truthed and refined through field sampling.

Key words: habitat capability, habitat suitability, mapping.

For the past 30 years, the province of British Columbia has been mapping wildlife habitat that is based on ecological mapping (Blower 1973, Demarchi et al. 1983, RIC 1999). During that time the methodology evolved from mapping ungulate winter range capability at a single scale to applying habitat capability and suitability ratings for a diversity of species at scales from 1:250,000 to 1:20,000. Wildlife species for which habitat maps are most commonly produced include bears, ungulates, and species at risk. The criteria for developing and applying the wildlife habitat ratings have been standardized to maintain reliability and consistency across the province. This paper discusses the standard definitions, habitat rating criteria and procedures outlined in *BC Wildlife Habitat Rating Standards* (RIC 1999). Anyone undertaking a habitat capability and suitability assessment project should read the standards document in its entirety and complete the associated Resources Inventory Committee training course.

CAPABILITY AND SUITABILITY RATINGS

A habitat rating is the value assigned to a habitat for its potential to support a particular species of wildlife. There are 2 aspects to habitat ratings: suitability and capability. Suitability is defined as the ability of the habitat *in its current condition* to provide the life requisites of a species. Capability is defined as the ability of the habitat, under optimal natural conditions to provide life requisites of a species—irrespective of its current habitat conditions.

Terrestrial Ecosystem Mapping uses a 3-level classification hierarchy of ecological units (RIC 1998, Demarchi et al. 2000). Ecoregion and biogeoclimatic units are broad-level delineations portrayed on provincial-level maps. Within these broader units, site-level polygons describe ecosystem units composed of site series, site modifiers, and structural stages (RIC 1998). The structural stage category describes the existing dominant stand appearance based primarily on stand age and species composition. The structural stages range from nonvegetated to old forests.

The predictive ability of terrestrial ecosystem mapping (at the site series level) is particularly valuable for describing the value of wildlife habitat; even if the current structural stage is not suitable for an animal species, it may have the capability to become suitable with successional changes in the plant community over time or through specific management activities.

Figure 1 shows the different structural stages of the Biogeoclimatic Ecosystem Classification system (Meidinger and Pojar 1991) used in Terrestrial Ecosystem Mapping. This illustration represents 1 particular site series or habitat over time. Using food and cover requirements for black-tailed deer in the winter as an example:

1. the shrub/herb stage would have a low suitability, due to lack of cover and available food under the snow; and
2. the older forest stages would have a high suitability for winter food and cover habitat, due to the deer's ability to walk on top of the snow to feed on litter fall.

Assume that at present this site is a clearcut that is several years old, currently in the shrub/herb stage (structural stage 3). The clearcut has a low suitability as deer winter habitat. However, this site does have a high capability,

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

because, given the right conditions (i.e., enough time), it will produce a forest that will provide deer winter habitat with a higher suitability. At the mature/old forest stages, if a particular site provides high quality deer winter habitat, then the suitability is equal to the capability of the site.

Density Measure

Animal density represents a (theoretical) measure for the value of habitat ratings. For the purposes of the capability/suitability ratings, density is: a species' response to a particular habitat expressed as the amount of time an animal may spend in the habitat within the season of use being evaluated (number of animals/unit time/unit area).

It is recognized that animal density measures are not always an accurate reflection of habitat quality. But the concept that the best habitats have the highest densities or the highest amount of use is still the guiding principle when applying wildlife habitat ratings in British Columbia.

The animal density measurement should ideally be based on actual measurement of animals in the habitat that is being evaluated, but this kind of data is rarely available. Thus, the densities used in the habitat ratings do not reflect actual or real numbers of animals. The animal density measure provides a conceptual framework for expressing expected use of a habitat.

Provincial Benchmark

The dictionary defines a rating as "a relative estimate or evaluation." A habitat suitability or capability rating is relative to the provincial benchmark, which is the highest capability habitat for the species in the province, against which all other habitats for that species are rated. Therefore, a capability or suitability rating is the potential value of a habitat to support a species, compared to the best habitat for the species in British Columbia.

Provincial habitat benchmarks standardize habitat values to ensure land-use decisions are based on an accepted methodology. They ensure that adjacent ecosystems are rated consistently and that habitat ratings are uniformly applied from 1 administrative unit to the next, from 1 mapping project to the next, and from 1 species to the next.

For example, the winter benchmark habitat for mountain caribou is in the Cariboo Mountains ecosection, in seral stage 6 of the Englemann Spruce–Subalpine Fir broad ecosystem unit. When rating winter habitat for mountain caribou you would rate it in comparison to the Englemann Spruce–Subalpine Fir ecosystem.

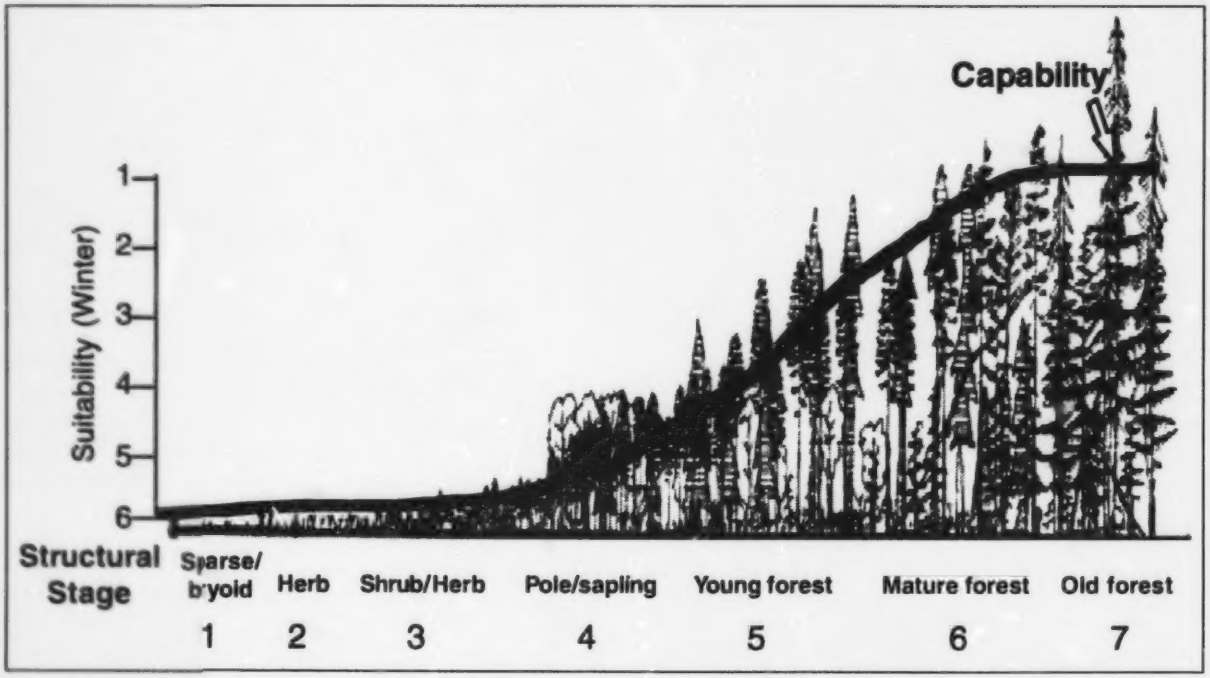

Figure 1. Habitat suitability for black-tailed deer in winter for a particular site over time. Suitability is the current ability of the habitat to support deer in winter. Capability is the ability of the habitat in its optimal condition (i.e., structural stage) to support deer in winter.

Table 1. Habitat capability and suitability rating schemes for 3 levels of knowledge about a species' use of habitat (RIC 1999).

% of provincial best	Substantial knowledge of habitat use (6-class)		Intermediate knowledge of habitat use (4-class)		Limited knowledge of habitat use (2-class)	
	Rating	Code	Rating	Code	Rating	Code
100–76	High	1	High	H	Habitat Useable	U
75–51	Moderately High	2	Moderate	M	"	"
50–26	Moderate	3	"	"	"	"
25–6	Low	4	Low	L	"	"
5–1	Very Low	5	"	"	Likely No Value	X
0	Nil	6	Nil	N	"	"

RATING CRITERIA

Planning needs and ecological mapping in the province range from the broad, strategic level (e.g., 1:250,000 map scale) down to a more detailed, operational level (e.g., 1:20,000 map scale). There are also varying levels of information available on species-habitat relationships for the diversity of wildlife species in the province. The standard methods for deriving habitat values from ecosystem maps require a great deal of flexibility to meet this range of map scales, information needs, and knowledge.

Rating Schemes

Three rating schemes have been developed to reflect the knowledge available on the species and how it relates to its habitat (Table 1). The appropriate rating scheme is the one that reflects our knowledge[1] of a given species' habitat use and the scale at which that knowledge is applied. The highest provincial rating is assigned to habitats of greatest concentration or importance to each species, based on previous measures or estimates in benchmark areas. All other habitats are evaluated in comparison to these benchmark habitats.

The 6-class scheme, used where there is a substantial knowledge of a species' habitat use, has ratings of high (1), moderately high (2), moderate (3), low (4), very low (5), and nil (6) for defined seasons and life requisites. This rating system is most useful at medium map scales (1:50,000 to 1:100,000) where seasonal habitat use can be readily depicted, and at large map scales (1:5,000 to 1:20,000) for many species where specific habitat uses can be identified.

The 4-class scheme is used for species for which there is an intermediate knowledge of habitat use. This scheme employs high (H), moderate (M), low (L), and nil (N) ratings for defined seasons and life requisites. The 4-class approach is most effective at medium (1:50,000 to 1:100,000) to small (1:250,000 to 1:500,000) map scales where seasonal information is available.

The 2-class scheme is used for species for which there is little information about their habitat use. Species in this category are assigned capability or suitability ratings of "habitat useable" (U) or "likely no value" (X). This scheme can be applied in most areas for almost any species, even with very limited information on habitat attributes, although the range of values is so broad that it is of limited value for species that we can map at greater detail.

Seasons of Use

When describing the seasons of habitat use, we can also go from the general to the detailed, depending on the species, map scale, and planning needs. For some species for which habitat use between seasons cannot be easily differentiated (e.g., most amphibian and reptile species), only 1 year-round season should be rated. At the other end of the scale, up to 6 seasons can be differentiated, such as early and late winter or spring.

Seasons can be described at 4 levels of detail (Table 2): 1-, 2-, 4-, and 6-season. Detailed seasonal information cannot be depicted for the broad habitat units that are shown at small map scales (e.g., 1:250,000). A 1- or 2-season rating must be used at these scales, regardless of the species. With larger map scales and a more detailed knowledge level of the species, seasonal use of habitats can be described in greater detail. A 4-season rating is then the minimum required. The 6-season rating subdivides winter and spring into early and late periods, a level of detail that is only used for grizzly bear, black bear, and ungulates. For example, skunk cabbage sites are very important as early spring habitat for bears, so it is often useful to rate this particular habitat use at larger map scales. Most commonly, though, only 2 seasons need to be rated: winter and growing. Note that the growing season is an amalgamation of spring, summer, and fall.

Life Requisites

Along with the season, ratings are applied to a particular life requisite (Table 3). Life requisites are the special

[1] Note that "knowledge level" refers to the provincial knowledge base on the species and, in particular, to the collective knowledge we have on how the species relates to its habitat, and what ecosystem attributes are important to the species.

Table 2. Four levels for describing seasons of habitat use (RIC 1999).

Level	Code	Description	Application
1-season	A	All seasons	• when habitat use between seasons cannot be differentiated (small map scales and/or species with low mobility)
2-season	W	Winter[a]	• when seasonal habitat use can only be roughly differentiated
	G	Growing (spring, summer, fall)	• small map scales
4-season	W	Winter[a]	• when 4 distinct seasons of habitat use can be differentiated (medium to large map scales)
	P	Spring	• species for which there is an intermediate or substantial knowledge level
	S	Summer	• when species occur in B.C. only part of the year (migratory species: only 3 of the 4 seasons rated)
	F	Fall	
6-season	WE	Early Winter	• when distinguishing detailed seasons for grizzly bear, black bear, and ungulates (for most of these species, only 4 or 5 of the 6 seasons will be rated; e.g., food habitat for bears may be rated for Early Spring, Late Spring, Summer, and Fall.)
	WL	Late Winter	
	PE	Early Spring	
	PL	Late Spring	
	S	Summer	
	F	Fall	

[a] Winter can be used for either a 2-season or a 4-season rating; in both cases, it is the same period of time.

requirements of an animal for sustaining and perpetuating the species. These requirements are supplied by the species' habitat, and include food, cover, reproduction, migration, hibernation, etc.

Generally, the food/cover life requisites can be rated separately, or combined into the more general "Living" life requisite. "Living" is the default for capability and suitability ratings. For more detail, the specific life requisites can be used to rate particular activities such as reproducing or hibernating.

Combining Seasons and Life Requisites

"All capability and suitability ratings are expressed as a value for a particular season and life requisite for the species of concern, compared to the best habitat in the province used for the same season and life requisite for that species." (RIC 1999)

With the numerous levels of detail in the rating criteria just described, there are many possible combinations of seasons and life requisites to rate. However, because the rating criteria depend on the animal species, the map scale, and the project

Table 3. Life requisites used in capability and suitability ratings (RIC 1999).

Life requisite	Code	Definition
FOOD/COVER		
Food	FD	habitat used for consuming food items, including searching for and consuming food simultaneously (such as done by grazers, browsers, flying insectivores, ducks, etc.)
Security	SH	habitat used for protection or hiding from predators
Security/Thermal	ST	habitat used for security and/or thermal values (this category used when differentiation between thermal and security values is difficult or impossible)
Thermal	TH	habitat used for protection from heat, cold, precipitation, or wind
SPECIFIC		
Courtship/Mating	CO	habitat used for courting, pair-bonding, or mating (when separate from reproducing habitat)
Living	LI	habitat used for general living activities
Migrating (seasonally)	MS	habitat used for regular, annual travel (e.g., habitat used by elk for spring and fall migrations)
Reproducing (birthing)	RB	habitat used specifically for giving birth to live young (mammals); may or may not include courtship/mating, depending on the animal species
Reproducing (eggs)	RE	habitat used for building a nest, laying eggs, incubation, hatching, and feeding non-mobile young (amphibians, birds, and reptiles); may or may not include courtship/mating, depending on the animal species
Staging	SG	habitat used for staging during spring and fall migrations

Table 4. Wildlife habitat rating criteria for a diversity of wildlife species at 3 map scales, showing rating scheme (RS; 2-, 4-, or 6-class), and minimum required and optional habitat use (adapted from RIC 1999).

Species	1:250,000		1:50,000			1:20,000		
	RS	Min. required habitat use	RS	Min. required habitat use	Optional habitat habitat use	RS	Min. required habitat use	Optional habitat use
tiger salamander	2	Living -All seasons	4	Living -All seasons	Living -Winter -Growing Reproducing	4	Living -All seasons	Living -Winter -Growing Reproducing
sharptail snake	2	Living -All seasons	2	Living -All seasons		4	Living -Growing Hibernating	
spotted owl	2	Living -All seasons	4	Living -All seasons	Living -Winter -Growing Reproducing	4	Living -All seasons	Living -Winter -Growing Reproducing
marbled murrelet	2	Reproducing	4	Reproducing		4	Reproducing	
Pacific water shrew	2	Living -All seasons	2	Living -All seasons		4	Living -All seasons	
pallid bat	2	Living -All seasons	4	Living -Growing	Hibernating Reproducing	4	Living -Growing	Food Security/Thermal -Growing Hibernating Reproducing
Vancouver Island marmot	2	Living -All seasons	4	Living -Growing	Hibernating	4	Living - Growing	Hibernating
caribou	6	Living -Winter -Growing	6	Living -Winter Growing	Food -Winter -Growing -Early Winter -Late Winter Security/Thermal -Winter -Growing	6	Living -Winter -Growing	Food -Winter Growing -Early Winter -Late Winter -Early Spring Security/Thermal -Winter -Growing Reproducing

objectives, the possibilities can be narrowed down to a few minimum requirements for particular species or species groups. Minimum requirements have been developed for ungulates, bears, and "Identified Wildlife" for 3 map scales (RIC 1999). Table 4 provides an example of the minimum required rating criteria for a number of species at risk.

SPECIES-HABITAT MODELS

For every species that is mapped, for every project, a Species-Habitat Model must be developed (or an existing model may be revised for the new project). It is particularly important to document the information on which the habitat ratings are based so that other biologists can assess and replicate the model.

Every model includes a written species account and a ratings table. The species account describes the species' habitat needs and the ecosystem attributes that supply those needs. It identifies the rating criteria and all the assumptions that go into the ratings table. The ratings table provides habitat ratings for all the unique ecosystems in the project area (i.e., it is the "look-up" table used with a GIS [global information system] to produce a habitat map from an ecosystem map).

A species account and a preliminary ratings table is developed for each species being rated prior to field sampling. Field sampling is undertaken in conjunction with the terrestrial

Table 5. Part of a Columbian black-tailed deer ratings table for one unique ecosystem in an example project in an example project in the Lake Cowichan area, showing food ratings for 3 seasons (RIC 1999).

Ecosystem description					Wildlife habitat ratings		
Ecosection	BGC[a] zone	BGC subzone	Site series	Structural stage	Food - Winter	Food - Early Spring	Food - Growing
LIM	CWH	xm	HK	2 - Herb	5	2	3
Leeward	Coastal	very dry	Western	3 - Shrub/Herb	5	2	3
Island Mtns.	Western	maritime	Hemlock	4 - Pole/Sapling	3	3	4
	Hemlock		Douglas-fir	5 - Young Forest	3	3	4
			Kindbergia	6 - Mature Forest	1	5	4
				7 - Old Forest	1	5	4

[a] BGC = biogeoclimatic

ecosystem mapping field work in order to ground-truth the preliminary ratings for each ecosystem in the project area. With this information, the ratings are finalized in the final ratings table.

Table 5 shows part of a ratings table for a unique ecosystem in an example project in the Lake Cowichan area. The 5 columns on the left describe 1 ecosystem in the project area—repeated for each structural stage. The 3 columns on the right are food ratings for Columbian black-tailed deer during 3 seasons. Compare the winter food ratings to the structural stage diagram in Figure 1 described earlier, where suitability for deer in winter increases with the age of the stand. Structural stages 6 and 7 (mature forests and old forests) are rated high (class 2) and very high (class 1), while structural stages 2 and 3 (the herb and shrub stages) are rated nil (class 5) to very low (class 5). Structural stage 7 is the capability for this site.

FURTHER INFORMATION

Additional information on wildlife habitat ratings criteria and the procedures for developing the ratings are provided in *BC Wildlife Habitat Rating Standards* (RIC 1999). This manual can be accessed at:
http://www.elp.gov.bc.ca/rib/wis/wildinterps. Printed copies can be purchased from the Queen's Printer in Victoria, B.C.

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A Case Study of Applied Habitat Suitability Assessment in Forest Management Planning

Amit Saxena
Geowest Environmental Consultants Ltd.
Suite 203, 4209 - 99 Street, Edmonton, AB, T6E 5V7, Canada
geowest@telusplanet.net

Kathleen Gazey
Slocan Forest Products Ltd.
Vavenby Division, Box 39, Vavenby, BC, V0E 3A0, Canada

ABSTRACT

As a result of Forest Development Round Table comment, a Wildlife Habitat Suitability Assessment was completed during the summer of 1998 in the Bone Creek area of the Clearwater Forest District, B.C. Habitat ratings were developed for grizzly bear (*Ursus arctos*), American marten (*Martes americana*), woodland caribou (*Rangifer tarandus*), and mountain goat (*Oreamnos americanus*). Significant habitats and connectivity corridors were identified for these species and included by Slocan Forest Products Ltd. in the context of the Forest Development Plan for the area. The Forest Development Plan was prepared based on a combination of: 1) General Resource Management Zone objectives, as outlined in the Kamloops Land and Resource Management Plan (LRMP); 2) landscape-level biodiversity and seral stage objectives, as identified in the Biodiversity Guidebook; and 3) significant wildlife habitats, as identified in the Wildlife Habitat Suitability Assessment.

Key words: caribou, forest management, grizzly bear, habitat suitability assessment, marten, *Martes americana*, mountain goat, *Oreamnos americanus*, *Rangifer tarandus*, *Ursus arctos*.

The legal framework for forest management on Crown lands in British Columbia is found in a combination of 4 provincial statutes: the Forest Act; the Forest Practices Code (FPC) of British Columbia Act; the Ministry of Forests Act; and the Range Act. In particular, the Forest Practices Code of British Columbia Act (1995) has established mandatory requirements for comprehensive ecological planning and environmentally sensitive forest practices. Through the FPC Act, more than 1,000 "landscape units" have been identified province-wide as the geographical framework within which sustainable forest management regimes should be implemented. Within these units, biodiversity objectives are identified for parameters such as forest seral stage distribution, temporal and spatial distribution of cutblocks, retention of old-growth forests, landscape connectivity, and forest stand structure and composition (Fenger 1996).

Slocan Forest Products Ltd., Vavenby Division, and Meeker Log and Timber, Clearwater Division, hold forest and partition licenses in the Bone Creek drainage within the Mud Landscape Unit of south-central British Columbia. In accordance with the Forest Practices Code, Slocan is required to implement actions compliant with regulations established by the FPC and to include its provisions in a 5-year Forest Development Plan. The forest development plan is an operational-level plan that sets the basic minimum requirements for forest management activities. Therefore, it must be tiered into, and incorporate the direction from, higher-level plans such as Land and Resource Management Plans (LRMPs). Figure 1 illustrates the context of a forest development plan, relative to other operational and strategic planning initiatives in British Columbia.

Stand- and landscape-level biodiversity objectives for the Mud Landscape Unit have been developed in conjunction with regional Ministry of Forests personnel and set forth in the *Biodiversity Guidebook* (B.C. MOF 1995). However, the Ministry of Environment has further specified that wildlife habitat suitability also be determined for key species and incorporated by Slocan into the overall Forest Development Plan. The Kamloops LRMP specifically states that habitat needs for regionally important and Red- and Blue-listed species are to be provided for. This project addressed 4 such species in the Bone Creek drainage: grizzly bear (*Ursus arctos*), American marten (*Martes americana*), woodland caribou (*Rangifer tarandus*), and mountain goat (*Oreamnos americanus*). The overall goal of this project was to provide a detailed account of habitat suitability and availability for these species and to address their habitat needs in the Forest Development Plan through stand- and landscape-level management.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

STUDY AREA

The Bone Creek study area consists of approximately 3,000 ha of isolated wilderness in the Mud Landscape Unit of south-central British Columbia. The study area is located northeast of the town of Blue River and west of Kinbasket Lake. It includes the upper reaches of Bone, Bannock, and Pancake creeks, all of which are second and third order tributaries of the North Thompson River.

The study area is generally steep-sloped with non-sorted colluvial materials being the dominant surficial materials in the drainage (O'Leary 1995). These materials vary in thickness from veneers on upper slopes to blankets on lower slopes, and are generally found on the steep valley walls in

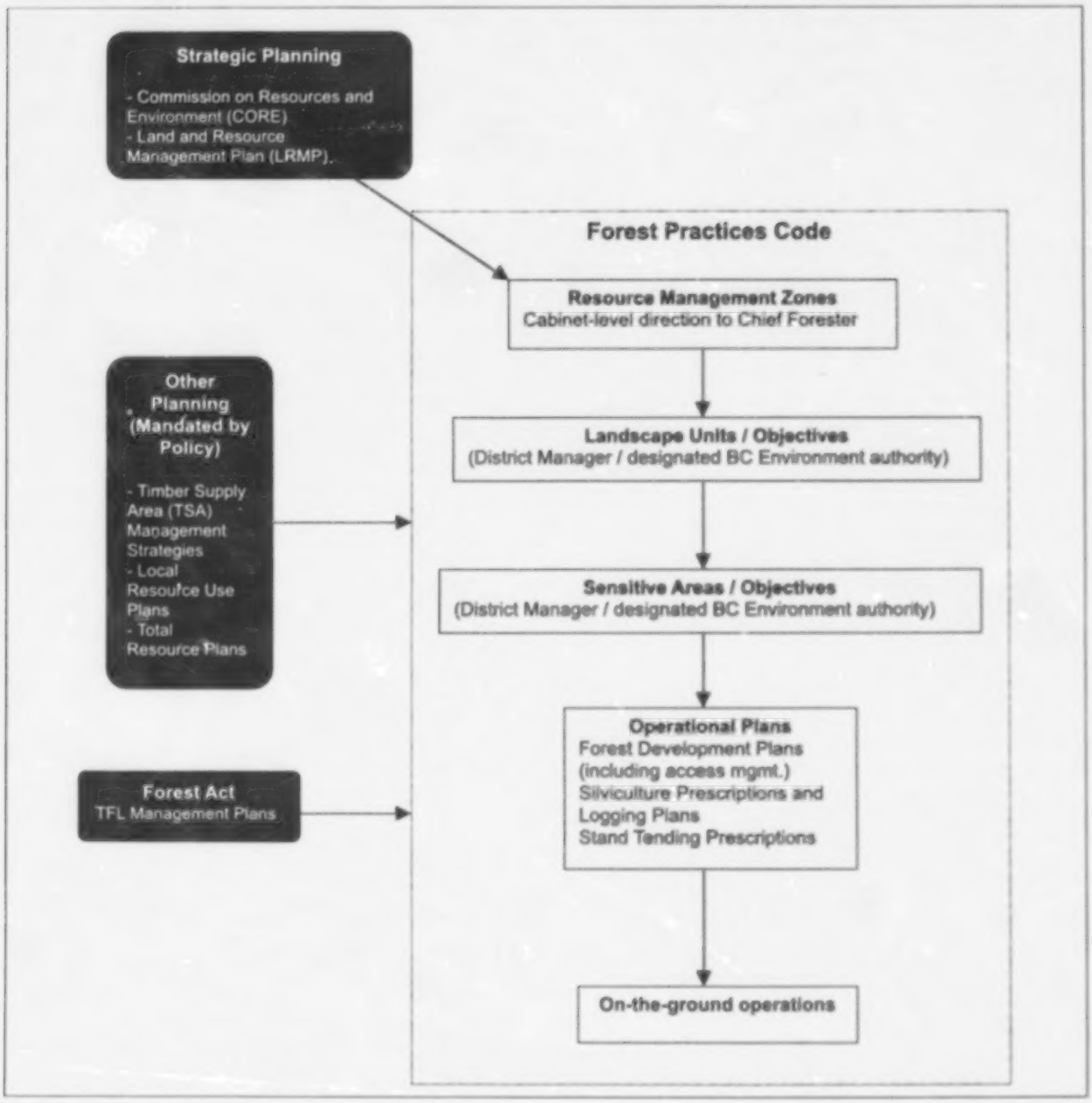
Strategic Planning
- Commission on Resources and Environment (CORE)
- Land and Resource Management Plan (LRMP).
Forest Practices Code
Resource Management Zones
Cabinet-level direction to Chief Forester
Other Planning (Mandated by Policy)
- Timber Supply Area (TSA) Management Strategies
- Local Resource Use Plans
- Total Resource Plans
Landscape Units / Objectives
(District Manager / designated BC Environment authority)
Sensitive Areas / Objectives
(District Manager / designated BC Environment authority)
Operational Plans
Forest Development Plans (including access mgmt.)
Silviculture Prescriptions and Logging Plans
Stand Tending Prescriptions
Forest Act
TFL Management Plans
On-the-ground operations

Figure 1. Planning levels in British Columbia (from Fenger 1996).

the form of cones and on lower-slope positions in the form of fans. Morainal materials are limited in the study area, but are found in the valley bottoms where they are commonly overlain with either fluvial or colluvial fans and aprons.

The study area spans 2 biogeoclimatic subzones: the Mica very wet cool Interior Cedar–Hemlock variant (ICHvk1); and the Northern Monashee wet cold Engelmann Spruce–Subalpine Fir variant (ESSFwc2). Within the drainage basin, the ICHvk1 variant reaches a maximum elevation of approximately 1,300 m, and is found throughout most of the valley bottom and lower-slope positions. Western redcedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) dominate mature climax forests in the ICH; however white spruce (*Picea glauca*), Engelmann spruce (*P. engelmannii*), hybrid spruce, and subalpine fir (*Abies lasiocarpa*) are common and can also form climax stands with the aforementioned species, especially where cold air drainages exist.

The mid- to upper-slope areas (above 1,300 m) are typically dominated by ESSFwc2 habitats. Engelmann spruce and subalpine fir are the dominant climax tree species in this portion of the study area. Spruce, which is typically the longer-lived species, usually dominates the canopy of mature stands, whereas subalpine fir is most abundant in the understory. At higher elevations within the ESSF, and in some wetter areas, subalpine fir frequently dominates the forest canopy.

Snow avalanche tracks are very common in this high snowfall, mountainous environment. These tracks, some of which transect both biogeoclimatic subzones of the study area, usually have a distinctive vegetation that is a tangle of tall shrub and herbaceous species, including slide alder (*Alnus crispa* ssp. *sinuata*).

The isolated and rugged nature of the study area has resulted in limited land use within the drainage. It is located within the Kamloops Land and Resource Management Plan, which has identified 6 Resource Management Zone categories for the LRMP area. The Bone Creek drainage lies entirely within the General Resource Management Zone, and therefore is subject to a basic set of objectives and strategies that guide land, water, ecosystem, and resource management. No portion of the drainage is designated within the Protection, Community Watershed, or Habitat/Wildlife zones; however, the extreme east portion of Bannock Creek is designated for Remote Access Recreation. Apart from timber harvesting, the only other significant land uses include heli-skiing, mountaineering, and hunting.

METHODS

In order to assign suitability values to habitats for the 4 featured species in this project, it was first necessary to classify the land base into discrete habitat units of like composition. In British Columbia, standards for terrestrial ecosystem mapping have been developed by the Resources Inventory Committee (RIC; RIC 1998*a*), whose objective it is to provide integrated standards for all resource inventories in the province. Concepts from the provincial Terrestrial Ecosystem Mapping (TEM) methodology and associated *British Columbia Wildlife Habitat Rating Standards* (RIC 1998*b*) were applied to the Bone Creek study area because we felt that the use of established provincial protocols would increase the utility of the product. The TEM program currently used in the province has evolved through the melding of 2 previous systems: the biogeoclimatic classification of the British Columbia Ministry of Forests (Mitchell et al. 1989), and the ecoregion classification of the British Columbia Ministry of Environment, Lands and Parks (Demarchi et al. 1990).

TEM-based concepts used in this project included the mapping of a 3-level system of hierarchically nested ecological units ranging from broad delineations of ecoregion units and biogeoclimatic units to site-level polygons describing specific ecosystem units. Other project components that were based on TEM standards described in RIC (1998*a*) were the field sampling protocol/intensity and the final polygon labelling and attribute coding. Although TEM-based concepts were used for development of the biophysical habitat unit map, this was not a TEM project. In particular, it did not entail the level of iterative correlation and reclassification that typifies most formal TEM projects.

Field sampling and verification were undertaken after preliminary pre-typing and stratification. Field work was completed from 1–8 August 1998, during which time 40 sample plots were visited. Field sampling was used to develop and refine the classification of ecosystem units and to confirm map unit designations and boundaries. It also increased the accuracy with which the resulting classification could be extrapolated and applied to portions of the study area not sampled. Information from the field plots also clarified interrelationships between plant communities, surficial materials, and soils, and allowed site-specific correlations between these ecosystem components and potential wildlife use.

The final biophysical habitat unit map was classified to the site series level and, subsequently, each polygon was assigned a habitat suitability rating for each of the 4 featured species. Habitat suitability was defined as "the current ability of the habitat unit to provide a wildlife species with the environmental conditions needed for cover, food, and space." Provincial standards and procedures for habitat suitability mapping are described in RIC (1998*b*) and were followed for this project. The following key elements guided the RIC-approved method of wildlife suitability rating for the project (see RIC 1998*b* and other papers in this volume for a detailed account of this methodology):

1. Alpha-numeric ratings were provided for each species in 1 of 3 rating schemes, each reflecting the provincial level of knowledge about the species being rated. Rating schemes

were described for those species for which we have detailed knowledge (6-class rating scheme), intermediate knowledge (4-class rating scheme), and limited knowledge (2-class rating scheme). The appropriate rating scheme was the one that best reflected the scientific community's knowledge of a given species' habitat use and the scale at which that knowledge was being applied.

2. These ratings were assigned as a percentage of the provincial benchmark. The benchmark is the highest suitability habitat for the species in the province, against which all other habitats for that species must be rated. The benchmark was used to calibrate the suitability ratings by providing the "standard" for comparing and rating each habitat or ecosystem unit. Note that the benchmark was an actual location, not a theoretical habitat.
3. Animal density measures were used as the conceptual framework for evaluating the value of a given habitat unit. The unit of measure was the amount of time an animal could spend in a habitat (number of animals/unit time/area of habitat). The "unit time" and "area of habitat" components of this equation are species-dependent (i.e., species mobility and scale of mapping affected each of these components). Note that there was no required calculation of actual density.
4. How an animal uses a particular habitat unit is closely associated with the season or time of year, and the specific activity or life requisite. Therefore, all ratings were expressed as a value for a specified season and activity, or life requisite.
5. A habitat suitability rating was provided for each species for every ecosystem unit (i.e., every site series/structural stage combination).

Summarizing the aforementioned criteria, a habitat suitability rating was defined as "a value assigned to a habitat for its potential to support a particular species for a specified season and activity compared to the best habitat in the province used by that species for the same season and activity."

Species-habitat relationship models in this project were meant to integrate key life requisites of the species with the mapped habitat or ecosystem attributes. The habitat rating criteria described above were applied to the TEM-based ecosystem map units through a habitat suitability model, which had 2 primary components: 1) a species account that provided background information on species biology and habitat requirements for each life requisite and associated season of use; and 2) a habitat suitability rating scheme and table that related the habitat requirements described in the species account to the relevant ecosystem attributes and ecosystem units described on the map.

When suitability ratings were applied to all polygons within the study area, a themed map of habitat suitability was produced for each species. An overlay of these habitat values for each species onto a single map resulted in the amalgamated portrayal of core wildlife habitat areas that were incorporated by Slocan into the Forest Development Plan.

RESULTS

Four species models and accompanying maps were developed for the study area (Saxena et al. 1998). Habitat values and suitability ratings determined for each of the 4 species in the study area were based on assumptions regarding the species' habitat ecology. These assumptions and associated habitat ratings in the study area are detailed in the species-habitat relationship models completed for this project (Saxena et al. 1998). The habitat values delineated for biophysical habitat units in the Bone Creek drainage are summarized below.

Grizzly Bear

Grizzly bear are afforded considerable high-rated (Class 1 and/or 2) spring habitat, which has been identified as the species' most limiting life requisite for habitat modelling purposes (Saxena et al. 1998). Grizzly bears descend to low-elevation wetlands and open mesic forests to forage on succulent, early green-up species such as horsetails (*Equisetum* spp.), sedges (*Carex* spp.), fireweed (*Epilobium* spp.), and cow parsnip (*Heracleum lanatum*). Numerous site series in the ICHvk1 subzone containing these types of forage items were rated Class 1, 2, or 3. Class 3 spring habitats were also found to be abundant in shrub/herb stages of low-elevation forests in the ESSFwc2.

Late summer and fall foraging habitats are also extremely critical and are satisfied through the provision of high-energy forage such as berries (*Vaccinium* spp., *Sheperdia canadensis*, *Arctostaphylos uva-ursi*) and starchy tubers. These habitats are likely to be found in higher-elevation areas adjacent to the study area. Avalanche chutes, particularly those on south-facing slopes, provide abundant forage and linear corridors between low- and high-elevation habitats.

Denning activity was expected to be concentrated at higher elevations. Only 2 mapped units within the study area were rated as providing the insulative snow cover and associated seepage required for denning habitat. The highest rated winter habitats were rated as Class 2.

Woodland Caribou

Some ecosystem units in the drainage have been rated as high as Class 2 for caribou, largely for the provision of spring forage. Low- to mid-elevation wetlands and other seral communities in the ICHvk1 were considered to provide early season forage. Forested wetland units and gently sloping avalanche chutes in the ESSFwc2 subzone provided spring forage for resident high-elevation portions of the caribou population.

Although small amounts of Class 2 winter habitats were identified, most of the caribou wintering range in the region is

located west of the North Thompson River in high-elevation forested habitats adjacent to Wells Gray Provincial Park (Stevenson and Hatler 1985). Snow conditions in the study area are anticipated to bury early winter shrub forage, while snowpacks remain too shallow to allow arboreal lichen foraging. By late winter, most caribou shift to higher-elevation subalpine and parkland habitats to forage exclusively on *Bryoria* spp. lichens. *Bryoria* spp. lichens are the preferred winter forage for caribou, but have limited occurrence in the study area. Lichen loads in the drainage are dominated by *Alectoria sarmentosa*, which is not as palatable as *Bryoria* spp.

Spring calving and summer foraging habitat were not identified in the study area. Upper-elevation subalpine parkland (ESSFp) and alpine tundra (AT) environments adjacent to the study area provide for these life requisites.

Marten

Mid-elevation mature to old-growth forests (structural stages 6 and 7) in combination with linear corridors of floodplain and riparian forest dominated by mature spruce provide the structurally diverse and mesic forest conditions required by marten. Where they occurred in large contiguous blocks, these habitats were assumed to provide all the life requisites, including denning/resting sites, overhead and ground cover, and prey species. These units were rated as Class 2 for marten.

Mature mesic forests provided the following habitat characteristics required by marten: 1) complex physical structure near the ground; 2) over-mature snags and cavity-laden trees; and 3) stand structure of various age and size classes. Such units were more common in the ESSFwc2, where wetter conditions fostered the growth of aforementioned conditions favoured by marten. Upland forests in the ICHvk1 are generally drier than similar forests in the ESSF and are expected to produce less woody debris, and a sparser and more homogenous understory, and therefore support lower prey densities than forests at slightly higher elevations.

Mountain Goat

Of the 4 featured species for which habitat suitability was rated in the Bone Creek drainage, the lowest suitability values and lowest availability of habitat were for mountain goat. The few forested cliffs and rock outcroppings in the project area were rated as high as Class 2 (mostly Class 3) for winter habitat use by mountain goats. Steep, south-facing slopes with mature timber stringers provide an appropriate combination of security and thermal cover under heavy snow conditions that force the goats to lower elevations. Under conditions of more moderate snow accumulation, higher-elevation alpine habitats outside of the study area may be utilized.

Core Wildlife Habitat Areas

Sixteen core wildlife habitat areas were identified in the study area, each encompassing a combination of habitat values for 1 or more of the project's featured species. The significant wildlife habitat areas were delineated based on an aggregation of habitat suitability values assigned to individual habitat units. These significant wildlife habitat areas were identified with a view towards integrating them with ongoing and future landscape-level planning initiatives, both within the Bone Creek drainage and adjacent to it.

DISCUSSION

The results of this project indicate that there are significant wildlife habitats within the Bone Creek drainage that must be considered in both fine-filtered and coarse-filtered resource management regimes. Through species-specific (fine-filtered) approaches to timber and forest management, the above-noted habitat values for each of the featured species within the Bone Creek drainage can be protected while maintaining access to valuable timber resources. As potentially significant habitats have been identified in the operating area, stand-level timber harvesting activities can be planned to consider these wildlife species. Results of the Wildlife Suitability Assessment directed Slocan to consider management strategies that would minimize forest fragmentation, while concomitantly maintaining natural landscape connectivity for wildlife in the area. Examples of timber management guidelines that were implemented by Slocan in the Forest Development Plan in order to meet the needs of the featured wildlife species included the following:

Woodland Caribou

1) maintain preharvest species composition and size class distribution; 2) maintain low evergreen shrubs in the ICH subzones for early winter forage; 3) maintain movement corridors with continuous windfirm bands of timber with naturally-pruned lower branches and canopy closure sufficient to intercept snow; 4) minimize visual obstruction; and 5) minimize access to the drainage.

Mountain Goat

1) limit access to kidding grounds during the May–June period (of limited concern in the study area because these habitats are generally located on cliffs with inaccessible ledges and are often in inoperable areas); 2) establish and maintain Wildlife Tree Patches on timber/rock outcrop complexes to provide thermal cover in ICH and lower ESSF winter ranges; and 3) encourage development of low shrubs by opening canopies in small patches where escape terrain is readily accessible.

Grizzly Bear

1) maintain cover in riparian zones, which are used for foraging and as travel corridors; 2) maintain buffers of at least

50 m around high-elevation meadows and avalanche tracks; 3) schedule harvesting operations and road building for times other than during spring habitat use in lower-elevation habitats; 4) winter logging activity will be limited in areas where high-rated denning habitat has been identified; however, in cases where ground conditions require snowpack or frozen ground, logging will be scheduled and completed as soon as possible after necessary ground conditions are met; and 5) implement Access Management Plans to limit access to the drainage immediately after completion of harvest and silvicultural operations.

MARTEN

1) retain an abundance of snags, rotting logs, and other coarse woody debris >7.5 cm diameter in retention patches within a partial cut system; 2) debris exceeding 50% ground cover will be avoided; 3) regeneration stocking targets will be considerate of marten preferences for intermediate stocking levels and intermediate levels of coniferous canopy closure (30–60%); 4) retain residual and retention patches with high shrub and forb diversity in understories; and 5) cutblocks in high-rated marten habitat should be widely dispersed, with leave patches of considerable size.

With the exception of marten, all other featured species in this project are large, highly vagile mammals. Characteristically, they tend to range over great distances and derive their life requirements from a diversity of habitats. Grizzly bear, caribou, and mountain goat use seasonally available habitats within the Bone Creek drainage, but also make use (sometimes extensively) of critical habitats adjacent to the project area (i.e., subalpine parkland and alpine habitats). The species that use habitats both within and adjacent to the Bone Creek study area, migrate seasonally between these areas as part of an annual cycle.

The need for connectivity among individual habitats within the study area (stand level) and between the study area and adjacent habitats (landscape level) is a reality that has been recognized and adopted by Slocan. As a matter of fact, environmental changes that preclude movement between component habitat patches may be as devastating to some mobile species as are the forces that directly destroy habitats and species (Frankel and Soule 1981). On the landscape-level, the habitat values and core significant habitats identified in this project provide the basic building blocks required for the design of an overall ecosystem management framework for the Landscape Unit.

MANAGEMENT IMPLICATIONS

A landscape comprises a mosaic of habitat elements (i.e., patches, corridors, and the intervening matrix). The scientific realms of landscape ecology (Forman and Godron 1986), biodiversity conservation (Wilson and Peter 1988), and conservation biology (Soule 1986) have emerged as trans-disciplinary sciences that attempt to explain the spatial patterns we see on and across the landscape: what they are, how they develop, how they change over time, and how they affect and are affected by ecological processes. Among anthropogenic activities that alter landscapes, timber harvesting is one of the most influential forces affecting the size and spatial arrangement of these landscape components.

Traditionally viewed as negative and deleterious to forest function, recent initiatives in landscape research from Oregon (Diaz and Apostol 1992), Washington State (Morrison and Swanson 1990, Shlisky 1993), Montana (O'Hara et al. 1993), and other areas of the Pacific Northwest (Ruggiero et al. 1991, Swanson et al. 1993, Shepard 1993) have elucidated the role of timber harvesting as a forest management tool to meet societal, ecological, and economic targets. We are witnessing a paradigmatic shift in forest management from sustained yield to long-term ecological sustainability. At the core of this new paradigm is the understanding, acceptance, and inclusion of natural disturbance as an integral component of ecosystem management. The guiding principle of this concept is the assumption that ecosystems have evolved under the influence of natural disturbance agents such as floods, drought, wildfire, windstorms, landslides, insect epidemics, and pathogens. These disturbance events have occurred in varying intensities (little to all trees destroyed), frequencies (once per year to once per 200 years), patterns (widespread to scattered to clumped), and sizes (<1 ha to >10,000 ha; McComb et al. 1994). The results of these historical forces are ecosystems that have evolved with a natural, or historical, range of variability. Forest management principles in British Columbia are shifting toward this ecosystem management paradigm, which dictates that landscape units be managed to mimic spatial and structural patterns within a historical range of variation (Hopwood 1991, Pojar et al. 1994, B.C. MOF 1995, Daigle and Dawson 1996, Eng 1997), albeit within the constraints of higher level plans such as LRMPs.

Landscapes that are managed to most closely resemble the natural range of variability of the ecosystem will provide resilient and healthy yet productive ecosystems. Wildlife species resident in these ecosystems have evolved and adapted in response to the disturbances and abiotic factors within it. Therefore, managing for ecosystems within some range of natural variability is also the most scientifically defensible method of sustaining habitat to maintain viable populations of native species (Swanson et al. 1993).

Multiple-core habitat patches connected by corridors are effective landscape-level tools to capture the full spectrum of the region's biodiversity, to include all centres of endemism and unique habitats, to maintain genetically distinct populations, and to guard against episodic extinctions (Soule and Simberloff 1986, Noss 1987). Stand-level

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999.

practices, such as maintaining wildlife trees, coarse woody debris, and understory vegetation diversity, will be addressed in forest development plans, silviculture prescriptions, stand management prescriptions, and fire management plans; however, stand-level applications derive their direction from higher-level plans such as landscape-level biodiversity objectives. Biodiversity objectives for the Mud Landscape Unit have been defined on the basis of desired seral stage distribution for ecosystems with a natural disturbance type characterized by rare stand-initiating events. Within both ICHvk1 and ESSFwc2 subzones, small openings <40 ha and moderate openings of 40–80 ha are below targeted ranges, while large openings on the order of 80–250 ha have exceeded targeted ranges by 226 ha in the ICH and 1,023 ha in the ESSF. Stand-level management for specific species will be implemented within this framework of higher-level biodiversity objectives.

Managers attempting to integrate wildlife habitat concerns with timber harvesting are faced with a multi-dimensional task. This integration must be considered both in long-term strategic planning and in short-run operational planning. The end result is that forest licensees will have to adopt a coarse-filtered approach to managing biodiversity on a landscape scale by maintaining a wide (and historically occurring) diversity of habitats that should support a wide (and historically occurring) diversity of wildlife species. Concurrent with this coarse-filtered approach is the necessity to manage specifically for endangered, threatened, or regionally significant wildlife species, whose needs may not be adequately met through the coarse-filtered landscape management regime. Species-habitat relationship modelling, and subsequent habitat suitability assessment, has proven to be an efficient method implemented by forest licensees to incorporate wildlife habitats within regional biodiversity objectives in British Columbia (Andrusiak et al. 1996; Saxena et al. 1998; Saxena and Bilyk 1998*a*,*b*; Bilyk and Saxena in prog.).

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Great Expectations: Attempting to Fill the Wildlife Inventory Data Gap

Lorne D. McIntosh[1]

Resources Inventory Branch, British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada

Key words: database, wildlife inventory.

EXPANDED ABSTRACT

Recent initiatives by our provincial and federal governments have raised public expectations that species at risk can and will be managed in a way that will improve their status. Expectation that such management will work seems based on several assumptions: external factors creating the risk can be identified; the party responsible for management has control over those factors; the manager has good understanding of demographics of the species being managed; the manager has good understanding of linkages between risk factors and affected species.

Assuming that ad hoc observations, inventory data, interpretations, hypotheses testing, knowledge, and understanding are incremental steps along a sort of continuum, then management action would most likely be well informed, and successful, if delayed until understanding of all relevant factors were reached. This degree of serendipity doesn't often occur.

The obstacles are daunting. Consider only the problem of species inventory in British Columbia. Much of the considerable data collected over the past several decades cannot be integrated statistically. Little systematic inventory of many wildlife species has occurred. The agency responsible for wildlife has little control over much of their habitats. Research to determine habitat-to-species linkages is expensive and chronically underfunded. There is an underlying need for statistically useful data placed within a common structure of metadata.

Although Broad Ecosystem Inventory (BEI) mapping covers most of British Columbia at 1:250,000 scale, the level of detail available is best suited for large-area planning. In comparison, Terrestrial Ecosystem Mapping (TEM) provides much more detail, usually at 1:20,000 and 1:50,000 scales, but, unfortunately, covers only a small part of British Columbia. Contrasting with those habitat types of information, wildlife inventory data sets that are statistically comparable in both the temporal and spatial dimensions are very sparse. Thus, data on most species exist only at the level of a range map. For most species very little is known about relative abundance across ecoregions and even less about their absolute abundance. Many questions remain in understanding the linkages between habitats and their dependent species. This situation makes well-informed management decisions difficult to reach and others easy to challenge. Although filling the data gap is a Sisyphean task, notable progress has been achieved recently.

Since 1991, British Columbia's Ministry of Environment, Lands and Parks (MELP) has been designated the data custodian for the province's wildlife inventory data. The advent of the Forest Practices Code in 1995 advanced both recognition of the need for reliable resources information and MELP's data custodian role. By early 1997 the ministry's Resources Inventory Branch had been given responsibility for that function. Forest Renewal British Columbia (FRBC) funded headquarters infrastructure development and many regional inventory projects. Two major infrastructure tasks included completion of a suite of inventory manuals, which recommend methods for the statistically sound inventory of priority species, and creation of a large-capacity Species Inventory database (SPI) for the storage, analysis, and distribution of data collected using those approved methods or from other compatible sources. Progress under both initiatives is the subject of separate presentations (Lepp 2000, Quayle and Westereng 2000). Many wildlife inventory projects have been completed to the new standards. Data from these and a backlog of compatible earlier inventories will soon be placed on an Internet-accessible site for use by wildlife managers, other professionals, and the public. In spite of these efforts many data gaps remain.

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[1] Author retired. Alternate contact: James.Quayle@gems9.gov.bc.ca

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Methods or Madness? Developing Population Inventory Manuals for British Columbia's Biodiversity

James F. Quayle
Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada
James.Quayle@gems9.gov.bc.ca

Leah K. Westereng
Resources Inventory Branch
British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada

ABSTRACT

In 1992, the Elements Working Group began developing standards for the collection of population inventory data about elements of British Columbia's terrestrial biodiversity. As the result of meetings between an assortment of wildlife biologists, the province's biodiversity was divided into 32 element groupings, and inventory methods and standards were developed around each of these, as well as for 5 additional topics relevant to biodiversity inventory. Manuals in the series are available for free via the Internet, and provide an obvious resource for biologists studying species at risk. In addition, developing methods for such a wide array of species has highlighted some of the general limitations of wildlife inventory, particularly for species that are rare or endangered.

Key words: inventory, monitoring, populations, standards.

In 1992, the Elements Working Group (EWG) was formed to work as part of a larger body, the Resources Inventory Committee (RIC), towards the development of data collection standards for the government of British Columbia. The EWG is a group of wildlife biologists with the specific task of developing standards for the collection of data about elements of the province's terrestrial biodiversity. These standards are intended to support a vision whereby all provincial data on the abundance and distribution of wildlife populations will be collected using standard methods, documented using common sets of codes and definitions, and will be easily accessible to all users via a centralized corporate data system.

Setting standards for the collection of data on biological populations is a daunting task in any jurisdiction, but particularly so in British Columbia. Although standards for aquatic and marine elements of biodiversity are the domain of other working groups, there are still an impressive number of so-called "terrestrial" species within scope, including many species that occupy the interface between the terrestrial regime and aquatic or marine habitats, such as waterfowl, amphibians, and aquatic mammals. The EWG intends to develop standards for conducting inventory of all of these species' populations in all of the different habitats in which they occur. This encompasses 15 species of reptiles, 20 amphibians, 478 birds, and 87 mammals, as well as more than 2,200 different vascular plant species and unknown numbers of invertebrates (Douglas et al. 1998, Resources Inventory Committee 1998). Included among the vertebrates are 48 species that are threatened or endangered in the province (Red-listed), and an additional 56 considered vulnerable (Blue-listed).

APPROACH AND PROGRESS TO DATE

As a first step, it was necessary to try to render the wide array of biota in British Columbia into more manageable packages. This was accomplished through a series of meetings in which wildlife biologists from a variety of disciplines were gathered together and formed into 5 specialist discussion tables: birds, mammals, herptiles, invertebrates, and plants/rare ecosystems. Central among the tasks assigned to these tables was the identification of major element groups for which similar methods could be used to collect inventory data. Generally, the inclusion of elements within a group was to depend on similarities between life histories, resource use and/or range, and detectability. Each element group would be the subject of its own manual outlining inventory standards. Manuals would be written by "experts" who were thoroughly familiar with the species in the element group, after which they would be reviewed by experienced peers.

As a result of these discussions, a set of element groups for British Columbia's terrestrial biodiversity was defined (Table 1), with each group requiring a separate manual outlining recommendations for appropriate methodologies and data collection standards. Each manual for an element group was to contain a summary of natural history information relevant to conducting inventory of the species, as well as standards for personnel, equipment, and daily and seasonal timing. In

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Table 1. Element groupings and generic manuals proposed for development into standards for British Columbia, and progress to February 1999.

Broad grouping	Element groupings and generic manuals	Completed to date[a]
General	Species Inventory Fundamentals	version 2.0
	Vertebrates of British Columbia: Scientific and English Names	version 2.0
	Live Capture and Handling of Wild Mammals, Birds, Amphibians and Reptiles	version 2.0
	Wildlife Radio-telemetry	version 2.0
	Voucher Specimen Collection, Preparation, Identification and Storage Protocol	UD
Birds	Marsh Birds: Bitterns and Rails	version 2.0
	Colonial-nesting Freshwater Birds	version 2.0
	Nighthawks and Poorwills	version 2.0
	Marbled Murrelet	version 1.1
	Raptors	version 1.1
	Riverine Birds	version 2.0
	Shorebirds	version 1.1
	Forest and Grassland Songbirds	version 2.0
	Swallows and Swifts	version 2.0
	Upland Gamebirds	version 1.1
	Waterfowl	UD
	Woodpeckers	version 1.1
Mammals	Bats	version 2.0
	Bears	version 2.0
	Beaver and Muskrat	version 2.0
	Hare and Cottontails	version 2.0
	Marten and Weasels	version 2.0
	Medium-sized Territorial Carnivores	version 1.1
	Moles and Pocket Gopher	UD
	Mountain Beaver, Bushy-tailed Woodrat and Porcupine	version 2.0
	Pikas and Sciurids	version 2.0
	Small Mammals	version 2.0
	Ungulates: Aerial Inventories	version 1.1
	Ungulates: Ground-based Inventories	version 2.0
	Wolf and Cougar	version 2.0
Herptiles	Plethodontid Salamanders	version 2.0
	Pond-breeding Amphibians and Painted Turtle	version 2.0
	Snakes	version 2.0
	Tailed Frog and Pacific Giant Salamander	version 1.1
Other	Terrestrial Arthropods	version 2.0
	Macrofungi	version 1.1
	Rare Vascular Plants, Lichens and Bryophytes	UD

[a] UD = under development.

addition, standard methods were to be recommended at 3 levels of intensity: the presence/not detected level, providing the least rigorous data identifying the presence or failure to detect a species in a study area; the relative abundance level, providing indices to monitor changes in species abundance over space or time; and the absolute abundance level, providing data on the actual number or density of a species per unit area.

In addition to species-specific manuals, a small group of more generic standards were proposed, which would provide general guidelines for all population inventories, regardless of taxa. The first of these, *Species Inventory Fundamentals*, discusses general standards for survey design and provides guidelines for survey preparation, habitat description, and data analysis. *The Vertebrates of British Columbia* sets the taxonomic standard for the province, listing the conventional names for all species and subspecies of vertebrates in the province. *Live Capture and Handling* outlines ethical principles and logistical concerns for the capture, restraint, possible containment, and subsequent release of wild animals. *Wildlife Radio-telemetry* outlines generic principles of telemetry, such as types of equipment and study designs, and a literature review of different methods of transmitter attachment and their success on different species. Finally,

Voucher Specimens discusses requirements for the collection of vouchers, their preparation, storage, and transfer to a provincial custodian.

To date, manuals outlining inventory standards, including data forms, have been completed for 33 of the total proposed set of 37 (Table 1). All the current manuals in the *Standards for Components of British Columbia's Biodiversity* series are available to download at http://www.for.gov.bc.ca/ric/. From the very beginning, it was expected that the series would evolve as biologists "field-tested" the manuals in different areas of the province. Abundant feedback from different field workers has contributed to the improvement of many of the version 2.0 manuals, newly released in 1998.

DISCUSSION: SPECIES AT RISK

The inventory manuals in the *Standards for Components of British Columbia's Biodiversity* series provide an obvious resource for biologists conducting studies of species at risk. Each manual contains a selection of well-established methods for evaluating presence and/or abundance of different species, including guidelines to increase likelihood of detection. In addition, the development of this series of manuals has revealed some useful generalities about conducting inventory of wildlife species, which may be particularly relevant to a biologist studying species at risk. Two such generalities concern the importance of understanding limitations of inventory methods and the data they produce.

The first of these involves the limitations of the different levels of survey intensity, specifically the "Presence/Not Detected" level. Biologists attempting to document the range of a rare or endangered species may initially wish to utilize the methods described under "Presence/Not Detected." However, Boulanger and Krebs (1998) emphasize that non-replicated surveys following presence/not detected methods will not yield data of statistical value. Specifically, it is not possible to determine species absence or local extirpation without more structured survey methods. As this may be a crucial objective, especially for species at risk, biologists should consider upgrading methods to those used to evaluate relative abundance. These may not require much more effort, but will be more effective in determining species absence or local extinction if spatial and temporal replication is utilized, particularly for species that occur at very low densities. Surveys that use relative abundance methods also provide preliminary data for future surveys, which can be used to produce rough estimates of the number of samples required to detect a species in a specific habitat.

The second generality centres around study design and its effect on inference. In particular, 1 of the key issues that has emerged from the development of >30 species inventory manuals is the ability to derive habitat inferences from surveys that are generally focused on obtaining data about population numbers. Biologists studying species about which little is known (a common characteristic of species at risk) will likely be tempted to derive as much knowledge as possible from a limited amount of data. They are cautioned to remember some of the fundamental differences in survey design between habitat and population sampling. As an example, population surveys generally try to target periods of greatest detectability in the focal taxa and to shorten the duration of sampling to minimize fluctuations in population size, while habitat studies are more likely to time field work with limiting seasons or periods of specific interest, and to continue for longer durations to get a more complete picture of an animal's life history. Thus, it may be misleading to derive habitat use patterns from the "snapshot" obtained from a population inventory. However, this does not mean that it is impossible to design studies that will provide information about both population size and habitat use, only that biologists should be cautious about assuming every study affords this opportunity. Manly et al. (1993:5) discuss the limitations of making habitat inference at the population level, as would be the case for most population inventories. A well-conceived, dual-purpose study, although likely not capable of discriminating habitat selectivity, may still provide some information on habitat use where little or none existed before. Given the reality of shrinking funding for the inventory of all species, it is only sensible to try to make the most out of every new survey.

ACKNOWLEDGEMENTS

This project was possible only with the help of more than 100 professional biologists. Their contributions are greatly appreciated.

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British Columbia's Species Inventory Information System

Gerhardt Lepp

Resources Inventory Branch, British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria BC, V8W 9M1, Canada
Gerhardt.Lepp@gems8.gov.bc.ca

ABSTRACT

The Resources Inventory Committee (RIC) was established in 1991, with the primary task of establishing data collection standards for effective land management. This process involves evaluating data collection methods at different levels of detail and making recommendations for standardized protocols based on cost-effectiveness, cooperative data collection, broad application of results, and long-term relevance. The RIC process has defined the inventory methods and the data to be gathered by the Species Inventory system (SPI). The British Columbia Ministry of Environment, Lands and Parks, Resources Inventory Branch, Wildlife Inventory Section in Victoria is developing SPI. The application will be deployed over the Internet to wildlife inventory contractors, forest licensees, and Forest Ecosystem Specialists, who will access the application through a web page. After the data are entered, extracts of wildlife observation data can be downloaded for reporting and analysis. Government employees will access the application over the Wide Area Network. Public access may eventually be available to the data warehouse. SPI will facilitate: more complete and consistent data gathering and documentation for wildlife inventories; better quality assurance of data; access to data throughout the province subject to project and species-level security restrictions; support for the Forest Practice Code, wildlife management, and biodiversity initiatives; and assisting the British Columbia Conservation Data Centre (CDC) to catalogue and monitor rare and endangered species. SPI consists of a number of components, including on-line data capture, operational database, data migration, data warehouse, taxonomy database, geo-reference conversion, Winframe application server, and system security. A data-driven observation form is used to accommodate any number of observation forms used for different species groups and methodologies. The on-line data capture application is complete and went into production in the fall of 1998. Existing ungulate inventory data have been converted and loaded into SPI, and more current species inventory projects from across the province are currently being entered. Development of the operational database is almost complete.

Key words: database, wildlife inventory.

BACKGROUND

Recent government initiatives including CRII (Corporate Resource Inventory Initiative), CLIB (Common Land Information Base), and FRBC (Forest Renewal British Columbia) have provided an opportunity to improve the management of all renewable resources on provincially managed lands. CRII funding has largely been directed towards strategic or higher-level planning. CLIB funding has been directed towards acquiring information to facilitate First Nations' treaty negotiations. FRBC has allocated funds to support sustainable forest management efforts and implement the British Columbia Forest Practices Code (FPC), which requires that forest tenure holders manage specified wildlife, as well as the forest resources, on tenured lands. The collection of wildlife inventory data within the province is a major part of all 3 of these recent government initiatives.

With increased opportunities for wildlife inventory, the multi-agency Resources Inventory Committee (RIC) was established in 1991, with the primary task of establishing data collection standards for effective land management. This process involves evaluating data collection methods at different levels of detail and making recommendations for standardized protocols based on cost-effectiveness, cooperative data collection, broad application of results, and long-term relevance.

The Wildlife Program of the British Columbia Ministry of Environment, Lands and Parks (MELP) is the principal agency responsible for maintaining the diversity and abundance of native species and their habitats throughout the province. The need for wildlife inventory information is an identified key issue. Within the program, the Wildlife Branch and Resources Inventory Branch are responsible for coordination of species inventory activities. The British Columbia

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 456pp.

Conservation Data Centre (CDC) is responsible for cataloguing and monitoring rare and endangered species within the province. These provincial species inventories provide the initial indication of rare or endangered species and continued monitoring of population demographics over time.

The current manual processes of managing data are not adequate. Tens of millions of dollars worth of wildlife inventory data are being collected and stored on paper forms and spreadsheets. The information is difficult, expensive, and time-consuming to locate, obtain, and understand. Each spreadsheet is different, and quality assurance is not adequate. For example, UTM (universal transverse mercator) data can be collected in NAD27 datum or NAD83 datum. The datum of much of the data is unknown, which introduces large uncertainties in the data.

OBJECTIVES

The Species Inventory (SPI) system is intended to meet the following objectives:

GENERAL

- Support the Forest Practices Code.
- Support wildlife management and efforts to protect biodiversity.
- Facilitate research and education.
- Facilitate improvement of the quality of the data that are gathered.

SPECIFIC

- Implement standard inventory methods for components of British Columbia's biodiversity.
- Collect wildlife inventory data in a consistent manner and add value to the data. Figure 1 illustrates how value is added to the data by using the data in conjunction with other systems.
- Make data easier to access, understand, and use.

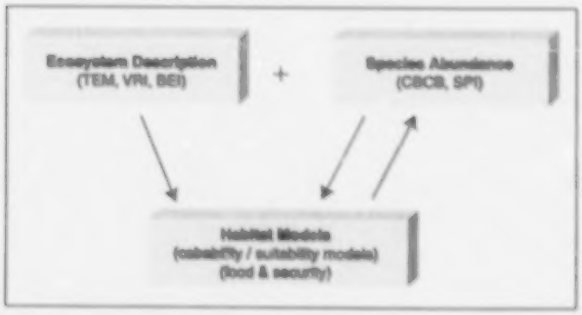

Figure 1. Wildlife Inventory Linkages.

STANDARD INVENTORY METHODS FOR COMPONENTS OF BRITISH COLUMBIA'S BIODIVERSITY

Wildlife inventory manuals are under development for approximately 36 different taxonomic groups in British Columbia; in addition, 6 supporting manuals will also be available. These are being compiled by the Elements Working Group of the Terrestrial Ecosystems Task Force, under the auspices of RIC. The objective of the working group is to develop inventory methods that will lead to the collection of comparable, defensible, and useful inventory and monitoring data for the species populations.

SPI REQUIREMENTS

SUPPORT RIC SPECIES INVENTORY METHODOLOGIES

About 36 species groups must be supported. This includes about 150 different observation forms with about 200 different columns. Many observation forms share common columns such as Activity, SEX, and Reproductive Condition.

PROVIDE ACCESS TO MANY USERS

- Inventory contractors/consultants: It is estimated that up to 100 contractors will require access to the system to enter inventory data.
- Forestry companies: There are more than 100 potential users in this sector that will require access to the database to answer questions that are relevant to their operations.
- MELP employees: It is anticipated that this will be our single biggest user group. MELP employees will access the system to monitor contracts, to perform QA analysis on their inventory contracts, and to analyze archived inventory data for wildlife management. Figure 2 illustrates the

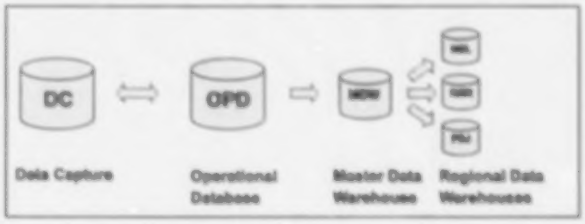

Figure 2. Components of the SPI database. The data are captured in a database that can be accessed via an application on a Winframe server via the Internet. After the contract monitor checks the quality of the data, he or she runs a procedure to move it to the operational database. The operational database is mostly read-only and the interface is optimized for browsing observation data. Frequently accessed views will be copied to the master data warehouse and the regional data warehouses.

components of the database that will facilitate different types of data access to different users.

- Forest Ecosystem Specialists (FES): The FESs have the most demanding requirements on the system, as they work in Ministry of Forests (MOF) offices. They will need access to the most sensitive data in the database in order to evaluate forestry plans and to enable them to provide well-informed recommendations to MOF.

Support Change

- The SPI system currently supports about 24 RIC methods with different versions of each method. Addition of new species inventory methods and versions of the methods is data driven and does not require modification of the database structure or the data capture application.
- It is recognized that methodologies will change. The system must use a design that accommodates all inventory methods and adapts easily to changes in methodologies.
- The system must minimize support costs because of the limited funding and the challenging requirements of the system.
- The system must support survey-specific observation columns. Many projects will need to record observation data that are not included in the RIC forms.
- The system must support security requirements and restrict access to Wildlife Observations and Wildlife Feature Observations for some projects with sensitive data.
- Rare and Endangered Specialists (RESs) are responsible for granting access to sensitive data. The system must allow the RES to selectively grant on-line access to data on a project-by-project basis.
- Contract monitors and/or project leaders should control basic data administration and security to minimize supports costs and allow users to control their own data.

COMMON PATTERNS IN WILDLIFE INVENTORY

It is not possible to build a species inventory system for a widely diverse group of species without assuming that there is a common pattern to all wildlife inventories. Figure 3 illustrates the design hierarchy that is common to all of the RIC species inventory methodologies. Figure 4 shows how this has been translated into a conceptual data model.

Types of Surveys

All surveys fall into 1 of the following categories:

- Absolute Abundance (AA): Calculation of the density of a population.
- Relative Abundance (RA): Index of abundance, which may be comparable across surveys.
- Present/Not Detected (PN): Indication of whether or not a species is present in the study area.

Common Questions

All surveys must answer a common set of questions:

What was observed?

The Taxonomy database is used to provide a consistent reference to the species that were observed, and to allow queries of observations of a particular species. Observations usually record other data about the time and place of the observation and the characteristics of the species observed.

How was the sample taken?

The SPI system provides the tools to document a statistically valid design with more rigour than can be accomplished in spreadsheets. The RIC species inventory method used must be recorded before observations can be recorded. The use of the RIC methods provides a sound structure for the survey.

What was the sampling effort?

A measure of the search effort is essential to provide the basis for Absolute Abundance calculations. A measure of the search effort is necessary to evaluate the confidence level of a Present/Not Detected survey.

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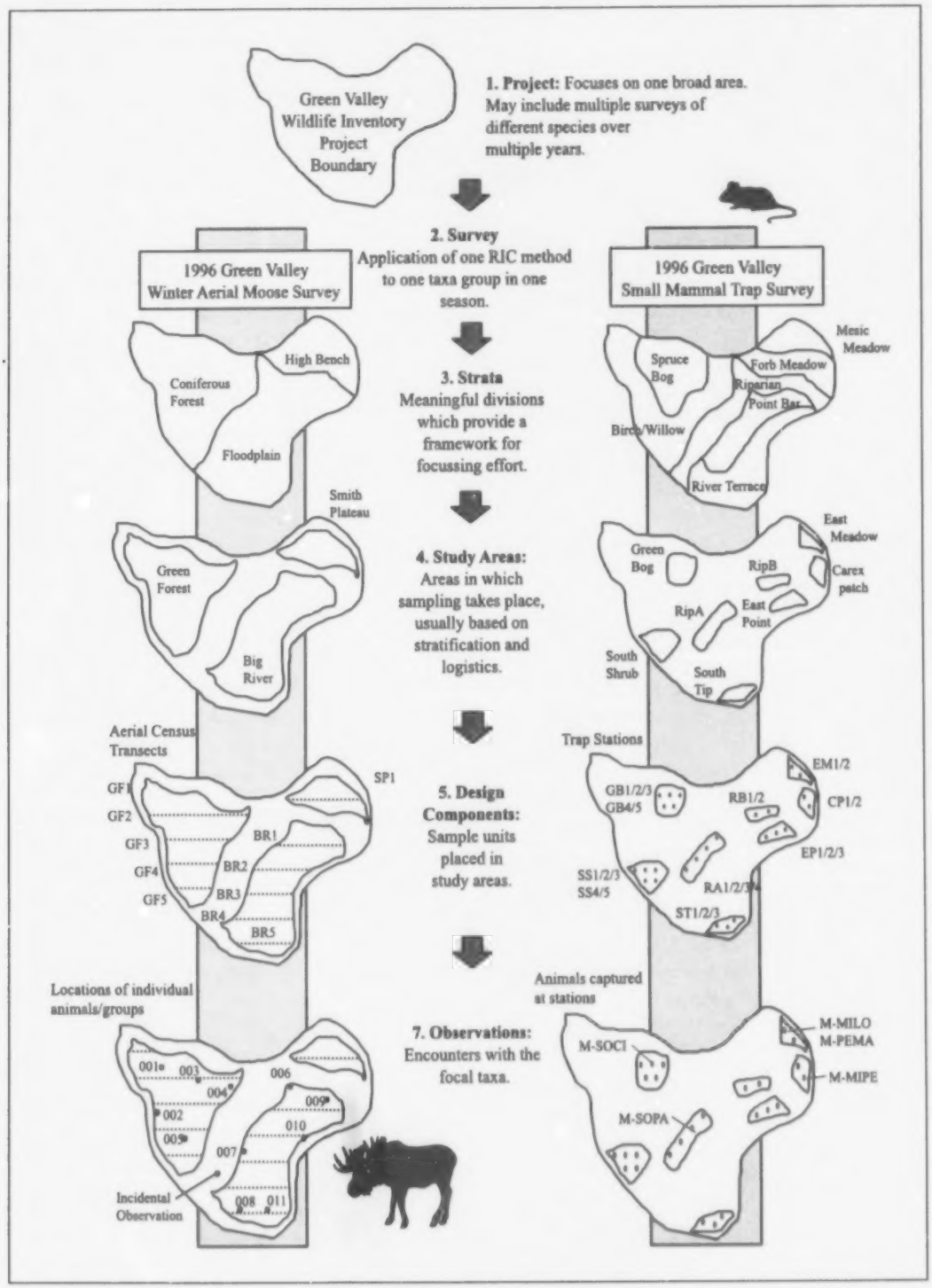

Figure 3. RIC species inventory study design hierarchy, with examples.

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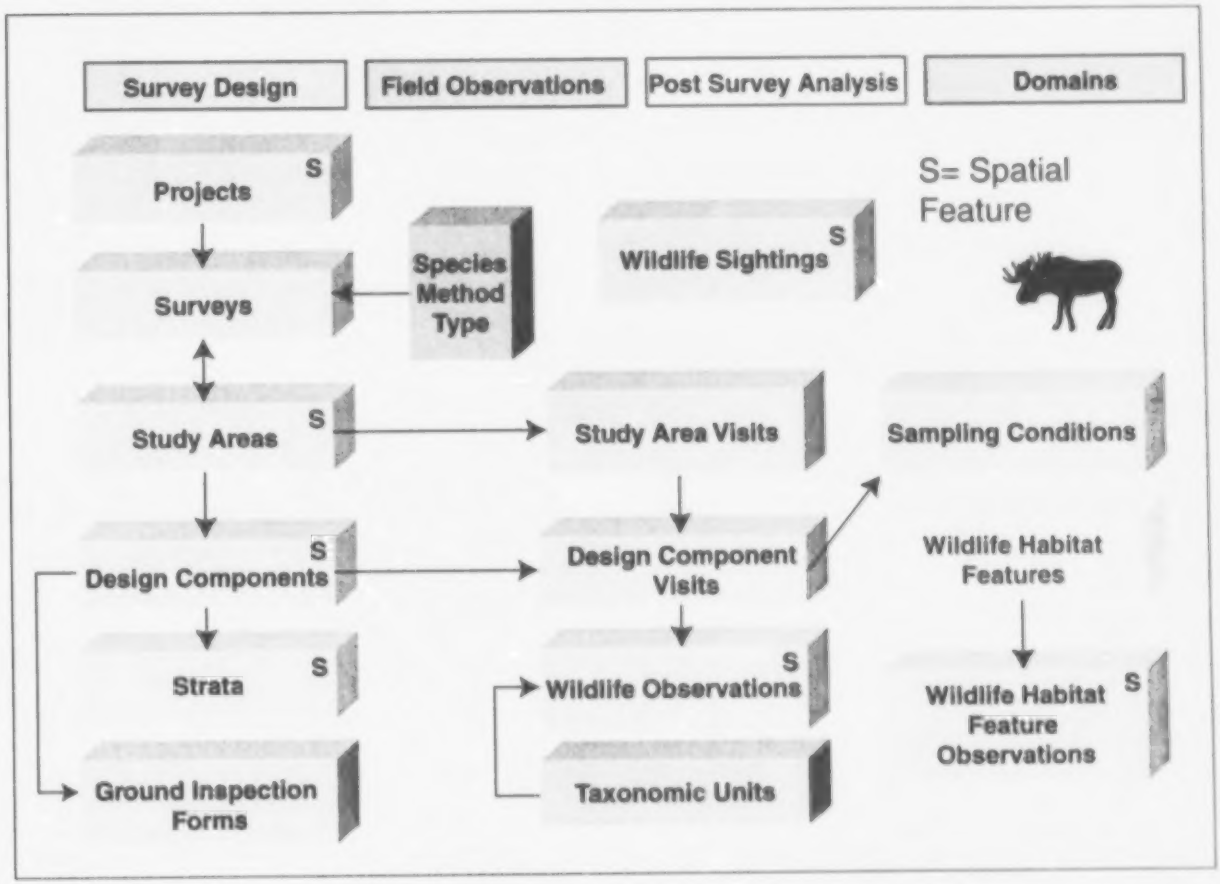

Figure 4. SPI Conceptual Data Model. This diagram includes the major entities or tables that make up the SPI database. An arrow with 1 head indicates a 1-to-many relationship. For example, each Project can consist of several Surveys, and a visit to a call station (Design Component Visit) could result in several woodpecker observations (Wildlife Observations).

How Useful is Automated Acoustic Recording in Amphibian Biomonitoring Programs?

Juanita M. Constible[1]

Department of Biology, University of Victoria
PO Box 3020 STN CSC, Victoria, BC, V8W 3N5, Canada

Key words: amphibian, automatic recorders, population surveys.

EXPANDED ABSTRACT

Amphibian declines and "normal" population fluctuations are easily confused if a study does not extend over a sufficient period. Resources for intensive long-term studies are typically not available, so managers and researchers have instituted volunteer amphibian monitoring programs to collect data. Although effective in densely populated regions, volunteer programs are less feasible in northern areas. In these cases, automated recording devices that do not require an investigator's presence may be a viable option.

I used automated acoustic recording devices (Dorcas and Peterson 1994, Kerfoot 1996), or "frog-loggers," for call surveys of frogs and toads at lakes in northeastern Alberta in 1997 and 1998. The units greatly reduced field time, but their use was problematic. Bears and other wildlife repeatedly disturbed the devices; I could not effectively assess numbers of calling males; and quiet or rare species were most likely underestimated or missed. The most appropriate use for automated recording units is to determine presence/not-found and diel and seasonal occurrence of calling behaviour. However, even for this kind of work, the devices are best used in conjunction with other survey techniques.

Based on my data, I make the following recommendations to managers interested in using automated acoustic recording in amphibian biomonitoring programs:

1. Train field staff well before the breeding season; mistakes early in the season result in costly errors and missed samples.
2. To facilitate data interpretation, use environmental data-loggers in conjunction with the automated acoustic recorders.
3. Reserve sufficient time and resources for cassette analysis.
4. In areas with dense black bear populations, take precautions to protect frog-loggers; field staff should have the equipment and expertise to repair microphones, and extra components should be available in case of damage.
5. Use more than 1 unit at a given water body to facilitate detection of rare or quiet species and to ensure activity patterns are accurately recorded; use high-quality microphones if the project budget permits.
6. Supplement frog-logger data with other survey methods, such as trapping or time-constrained searches.

Automated acoustic recording is potentially a useful tool for amphibian biomonitoring, especially in areas with small volunteer bases. However, the units have a relatively high start-up cost. Therefore, managers should carefully assess the likely utility of automated recorders before construction.

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[1] Present e-mail: jconstible@netscape.net

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Amphibian and Reptile Monitoring in British Columbia: Problems and Potential Benefits

C. Bishop
Canadian Wildlife Service, Environment Canada
P.O. Box 5050, 867 Lakeshore Road,
Burlington, ON, L7R 4A6, Canada
cab.bishop@ec.gc.ca,

S.A. Orchard[1]
Canadian Amphibian and Reptile Conservation Network

ABSTRACT

Monitoring for species diversity and relative abundance is a long-term proposition. It can also be a relatively expensive one. However, dependable government funding commitments to museums and public wildlife agencies that would be expected to carry out this work, have only diminished in the recent past. Consequently, today the resources are largely unavailable to adequately carry out this vital work in the long-term and on a large scale. Without the required data that defines species, populations and their relation to habitat, effective and comprehensive conservation efforts are next to impossible. In light of this, a trend has developed in Canada and elsewhere during the 1990s to rely less upon governments and to shift attention increasingly towards alternative sources of funding and manpower. Programs are now in place where specialists in amphibian and reptile biology enlist and co-ordinate teams of volunteers who conduct basic population surveys in and around their own communities. This system has proven its value in eastern Canada and on the Prairies. Survey forms are available for the various regions of British Columbia, but more money is needed before this or any other system can really produce effective results province-wide. More than any other part of Canada, British Columbia needs special attention because of its diversity of habitats and its specialized herpetofauna. In the main, the only obstacle to surmounting these unique challenges is bringing together the wit, the will and the means.

1 Present address: World Wide Fund for Nature Australia
GPO Box 528, Sydney, New South Wales, AUSTRALIA 2001

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One
B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Pheromone Trapping for Flying Beetle Biodiversity in Mature Northern Interior Douglas-Fir

Susanna Carson
University of British Columbia, Department of Zoology
6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada
carson@bc.sympatico.ca

Key words: *Dendroctonus pseudotsugae*, Douglas-fir beetle, pheromones.

EXPANDED ABSTRACT

Balancing species conservation and the maintenance of landscape-level processes has created a need to assess biodiversity and the impact of single-species management. In the Fort St. James Forest District, in central British Columbia, Douglas-fir grows at the northern extent of its range in North America. It comprises roughly 1% of timber stands in the district, and it is only found in scattered isolated areas. At the local level, Douglas-fir is a rare and highly managed species. In the early 1990s Douglas-fir beetle (*Dendroctonus pseudotsugae*) populations established themselves in isolated and inaccessible Douglas-fir stands. In 1992, increasing losses of Douglas-fir habitat coupled with extremely low success in Douglas-fir regeneration threatened the loss of mature Douglas-fir habitat throughout the district.

Between 1993 and 1998 a 5-year operational research study was initiated to assess flying beetle diversity associated with the management of epidemic Douglas-fir beetle populations. The study was a cooperative effort between researchers at the University of Calgary, the University of British Columbia, the British Columbia Ministry of Forests, and Forest Renewal British Columbia.

Beetle diversity was measured from pheromone baited and unbaited Lindgren funnel traps located in mature, beetle-attacked, leading Douglas-fir habitat. Pheromone traps were baited with known Douglas-fir beetle aggregation pheromones: MCOL (1-methylcyclohex-2-enol), and Frontalin (1,5-dimethyl-6,8-dioxabicyclo[3.2.1]-octane). Traps were collected weekly or semimonthly through the duration of the Douglas-fir beetle flight (April/May through August/September). Results of the study identified 150 species from 40 families. Rank abundance analyses indicate log normal species distributions for both pheromone-baited and control traps. Baited trapping yielded similar species richness but significantly increased dominance indices compared to control traps.

The results suggest that the effect of single-species management in this circumstance is not limited to the target organism, and that pheromone trapping can influence species composition at the community level.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

A Decision-Support Tool for Managing the Risk of Tuberculosis and Brucellosis Infection in Northern Canada

Jonathan A. Mitchell
Faculty of Environmental Design, University of Calgary
2500 University Drive N.W., Calgary, AB, T2N 1N4, Canada
jmitche@ucalgary.ca

C. Cormack Gates
Faculty of Environmental Design
University of Calgary
2500 University Drive N.W.,
Calgary, AB, T2N 1N4, Canada

Rick Rowell
AXYS Environmental Consulting Ltd.
Suite 600, 555 - 4th Avenue S.W.
Calgary, AB, T2P 3E7, Canada

Kevin Lloyd
AXYS Environmental Consulting Ltd.
Suite 600, 555 - 4th Avenue S.W.
Calgary, AB, T2P 3E7, Canada

ABSTRACT

The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) lists the wood bison (*Bison bison athabascae*) as Threatened (a species likely to become endangered if limiting factors are not reversed). Recovery efforts have been planned and implemented by the National Wood Bison Recovery Team (WBRT) and its partners. One of the greatest obstacles to the recovery of wood bison is the presence of introduced cattle diseases, specifically bovine tuberculosis and brucellosis, in some northern herds. Approximately 2,500 bison remain in infected herds, while the number of bison in noninfected free-ranging herds has increased to 4,700 (2,700 wood bison and 2,000 plains bison [*B. b. bison*]). In addition, the diseases are of grave economic concern to the burgeoning commercial bison industry in the region. While the agricultural industry and governments currently support the recovery of free-ranging herds of bison, support would diminish if the diseases spread to the healthy wild herds. In response to these concerns, the Animal, Plant and Food Health Risk Assessment Network (APFRAN) conducted a risk assessment at the request of the Canadian Bison Association. We have undertaken a project to add to the utility of the APFRAN model by incorporating biophysical factors and local and traditional knowledge of bison movements into the risk assessment. An approach called friction modelling will be used to define a resistance landscape for bison. The product will be expressed as a digital map of friction values, or spatial patterns of resistance to bison movements. The underlying invasion probability generated by the APFRAN model will be adjusted using a function of the friction value. The models and associated maps will serve as a decision-support tool for the development of management actions and guidelines to reduce the risk of infection of susceptible populations. This, in turn, will serve the continued recovery of wood bison in Canada.

Key words: *Bison bison*, brucellosis, risk assessment, tuberculosis, wood bison.

Wood Buffalo National Park (WBNP) and the surrounding area, located in the northeast corner of the province of Alberta and the southern Northwest Territories, supports the largest metapopulation of wood bison (*Bison bison athabascae*) in existence. Unfortunately, herds in this area are infected with 2 cattle diseases introduced during the 1920s, resulting from a massive release of infected plains bison (*B. b. bison*) imported from east-central Alberta. *Mycobacterium bovis* and *Brucella abortus* are the causative agents of the 2 reportable diseases, bovine tuberculosis and bovine brucellosis. The role of the diseases in the population ecology of free-ranging bison is the subject of current research programs in Wood Buffalo National Park (BRCP 1996) and Yellowstone National Park (NPS 1998). The pathobiology of each disease is considered similar in bison and cattle (Tessaro 1988). The main clinical features of brucellosis are a high incidence of abortion during the first pregnancy following infection (Davis et al. 1990, 1991), and a low incidence of bursitis leading to arthritis and reduced joint mobility (Tessaro 1988). Tuberculosis in bison and cattle is primarily a pulmonary disease, although any organ system may be affected. Advanced tuberculosis is generally fatal (Tessaro 1988). Owing to the importance of brucellosis and tuberculosis as zoonoses, these diseases have been the subject of intensive, long-term eradication programs in livestock

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populations in Canada and the United States. The Canadian livestock population was declared free of brucellosis in 1985. Tuberculosis continues to occur sporadically in livestock herds throughout Canada, but the incidence of the disease is extremely low. The last outbreak occurred in a cattle herd in Manitoba in 1997 (D. Scott, Canadian Food Inspection Agency, pers. comm.). Bison in the Greater Wood Buffalo National Park area represent the last known reservoir of these diseases in Canada.

MANAGEMENT ISSUES

The issue of disease management was the subject of a controversial proposal to eradicate infected bison in the region (EAP 1990). Although the government of Canada rejected the proposed action, concern about the persistence of the diseases has not diminished. Indeed, the reasons for concern have increased since 1990 in 2 main areas, namely the risk to the conservation of recovering disease-free, free-ranging bison, and the risk to a rapidly expanding commercial bison industry in the region.

Conservation

Since the termination of interventive bison management in WBNP and the adjacent Slave River Lowlands around 1970 (Carbyn et al. 1989), the Greater WBNP (GWBNP) bison population has declined by >80%, from 16,000 in 1971 to 2,500 in 1998 (Gates et al. in prog.). This stock represents a mix of original wood bison and introduced plains bison (van Zyll de Jong et al. 1995) and was the source of 3 salvage projects undertaken to establish brucellosis- and tuberculosis-free herds representing as close as possible the wood bison phenotype (Gates et al. in prog.). There have been no management initiatives undertaken to promote growth or recovery of infected herds since the 1960s. During the past 3 decades, 6 wild populations of noninfected bison have been reestablished in northern Canada, and now number approximately 4,700 bison in aggregate. They include 5 wood bison populations and 1 plains bison herd (Fig. 1). Should any of these herds become infected, management support would likely diminish as it has for existing infected populations. Managing the risk of infection of the herds nearest to WBNP is the focus of initiatives in the Northwest Territories and Alberta and is a concern for 3 herds in British Columbia (Harper and Gates 2000). The Yukon population is

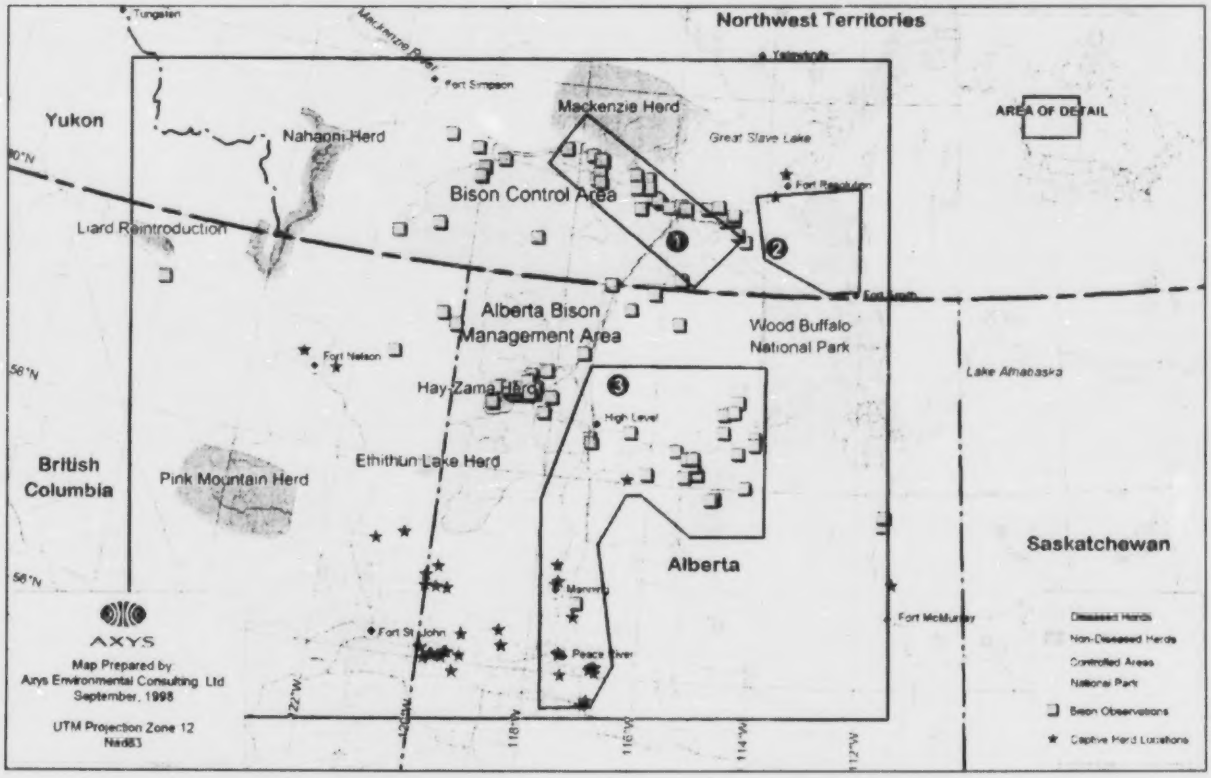

Figure 1. Disease risk management region illustrating 3 proposed study areas for disease risk modeling.

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sufficiently remote that diseases would only be of concern if closer herds became infected.

Other considerations in the management of diseased bison include the status of wood bison under COSEWIC (listed as Threatened), and the designation of WBNP by the United Nations Educational Scientific and Cultural Organization (UNESCO) as a World Heritage Site in 1983, predominantly because of wildlife populations that inhabit the park (Parks Canada 1997). A national program for conservation of wood bison has existed in various forms since the early 1970s. Currently, the Recovery of Nationally Endangered Wildlife (RENEW) program provides direction for a National Recovery Team, which provides advice to jurisdictions and coordinates recovery actions in Canada. This team has drafted a national recovery plan that identifies 4 primary goals to guide recovery efforts (Gates et al. in prog.):

1. Reestablish viable, healthy, free-roaming wood bison populations in original range, thereby contributing to the maintenance of ecological processes and biological diversity.
2. Ensure the genetic integrity of wood bison is maintained without further loss as a consequence of human intervention.
3. Restore healthy wood bison herds for long-term sustained use, thereby contributing to the aesthetic, cultural, economic, and social well-being of rural communities and society in general.
4. Encourage the establishment of long-term, cooperative management programs for wood bison, in which rural communities, aboriginal people, and public agencies play an integral role.

In addition, a Research Advisory Committee (RAC) was established in 1995 by the WBNP administration to provide advice for the park's Bison Research and Containment Program (BRCP 1996). The long-term goal of the program is to "maintain a viable, free-ranging bison population in the Park."

Commercial Bison

Production and Conventional Livestock

In the Peace River country of British Columbia, Alberta, and the Northwest Territories there are approximately 250 commercial bison herds that hold about 20,000 bison (D. Patten, Peace Country Bison Association, pers. comm.). The farms are located mainly in the agricultural zones in the 2 provinces. However, there are 2 farms in the Northwest Territories near WBNP, 1 of which is stocked with wood bison, and plans are being developed to stock the other. Two bison farms are located in the agricultural area west of WBNP in Alberta and another is proposed. The number of farms in the region has been increasing at a rate of 15% annually and the number of bison on farms has been increasing at a rate of 25% annually (Hussey 1997). Under the National Captive Ungulate Regulations, bison herds testing positive for either of the reportable diseases are subject to slaughter. The economic cost to producers of such an action is high, since losses due to condemnation are not compensated at full market value. Furthermore, access to national and international markets may be affected if a disease outbreak occurs. The Canadian Bison Association (CBA) formed a committee in 1996 to develop an approach to disease risk management. In 1997 the CBA asked the Animal, Plant and Food Health Risk Assessment Network (APFRAN) to conduct a risk assessment for disease-free, free-ranging bison, captive commercial bison, and cattle in the region around WBNP.

APFRAN'S DISEASE RISK ASSESSMENT

The risk assessment conducted by APFRAN (1998) estimated the probability of invasion by bison, contact, and disease transmission to an individual animal in each risk group, and the economic consequences of disease transmission. Risk was calculated by combining the probability of invasion with the probability of contact once invasion had occurred; then, contingent upon invasion and contact, the risk of transmission was factored in to define an overall risk of infection. Once that risk was defined, the assessment determined the economic consequences of infection.

The risk assessment for the year 1998 considered the 95% probability of at least 1 animal becoming infected in an "At Risk" group (Table 1). Free-ranging (wild) healthy bison were at the highest risk of infection, at 1 in 6 years for tuberculosis and 1 in 8 years for brucellosis.

A temporal model of infection probabilities for commercial bison was developed and projected 5 and 10 years into the future (APFRAN 1998). The model assumed that the rate of increase in the number of farms and the number of commercial bison would remain constant over the 10-year period. The risk of brucellosis infection increased to 1 in 64 years and the risk of tuberculosis infection increased to 1 in 44 years over the 10-year period (Fig. 2).

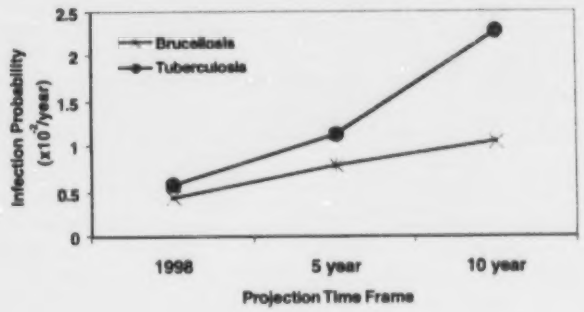

Figure 2. Temporal projections of the probability of infection for commercial captive bison (data from APFRAN 1998).

Table 1. Risk assessment results indicating infection probabilities and economic consequences based on conditions in 1998 (APFRAN 1998).

Disease	Risk parameter	Wild bison	Commercial bison	Cattle
	Probability of invasion	1/10 yrs[a]	1/29.9 yrs	1/2.7 yrs
Brucellosis	Probability of infection	1/8 yrs	1/229 yrs	1/1,276 yrs
	Cost per outbreak	$5,400,000	$6,500,000	$632,000
	Annual cost	$668,750	$28,384	$495
Tuberculosis	Probability of infection	1/6 yrs	1/173 yrs	1/1,764 yrs
	Cost per outbreak	$5,400,000	$8,200,000	$832,000
	Annual cost	$891,667	$47,399	$472

[a] Annual probability of 1 bison from GWBNP meeting a free-ranging bison from another population based on bi-directional movement.

Information Gaps

Bison movements and distribution are influenced by a number of biophysical characteristics of the landscape, such as habitat structure and mosaic pattern, topography, hydrography, physical barriers, and movement corridors such as cutlines, logging sites, trails, and roads. Management factors also influence distribution patterns, (e.g., bison are not protected from hunting outside of WBNP in Alberta, except in a designated wildlife management zone in the northwest corner of the province). The risk assessment conducted by APFRAN acknowledged the need for considering these factors; however, they were not included in the model due to "a lack of time and resources" (APFRAN 1998).

Local ecological knowledge (LEK) of bison movements and landscape features was not incorporated into the assessment and could be used to improve the application of the assessment for developing management recommendations. The area around WBNP is occupied by several First Nations, whose members still practice traditional subsistence, land-based lifestyles to some extent. Biologists, foresters, farmers, and hunters also possess knowledge of the area and could contribute information for interpreting likely bison distribution patterns.

Furthermore, bison are already present in low numbers in the area west of WBNP. The origin of these animals is unknown. They may originate from infected herds inside WBNP or from the disease-free, free-ranging Hay Zama herd in northwestern Alberta. Continued growth of the disease-free, free-ranging populations of bison was not considered in the risk assessment, nor was a scenario in which the disease-free herds became infected. The disease status of the free-ranging bison regularly reported in the area west of WBNP is poorly documented. Limited sampling (11 bison) detected *Brucella*-positive titres in 2 bison, and lesions, suspicious of tuberculosis, were seen in another bison (Tessaro 1988). Further testing of bison west of WBNP has been identified as a need by the Little Red River and Tall Cree First Nations.

These deficiencies were recognized by the CBA Disease Risk Management Committee (CBA–DRMC meeting, 6 October 1998, pers. comm.) and the Bison Research and Containment Program (BRCP)–Research Advisory Committee (RAC) for WBNP (D. Armstrong, RAC member, 1998, pers. comm.).

We have undertaken a project to address some of the deficiencies and to contribute to planning risk management guidelines and actions for the region. In particular, the project will develop a spatial model that incorporates biophysical factors from conventional data sources, such as digital remote sensing data bases, and from LEK sources, to define the resistance of the landscape to bison movements and distribution.

METHODOLOGY/PROJECT APPROACH

Development of a Digital Resistance Landscape

The risk assessment assumed equal distribution in every direction without biophysical or LEK considerations. We will incorporate biophysical and LEK data in a model that will define the resistance of the landscape to bison movements using a method known as friction modelling. This method has been used previously to model the potential for movement of wolves in the Banff and Bow Valley area of Alberta (Page et al. in prog.), and the movement of barren-ground caribou in the Northwest Territories (Gates et al. 1998). Friction modelling allows quantitative integration of biophysical and management factors that interact in dynamic and complex ways. Model development involves the following steps:

- identifying biophysical and management factors (variables) that influence bison movements through the landscape;
- measuring and digitally mapping the variables within the study area;
- developing and assigning significance weightings to each variable;
- assigning friction indices (ranging from 0 to 10) to the conditions described by each variable; and
- preparing mathematical statements to calculate an overall friction value from a unique set of friction indices for each pixel in the modelled landscape.

Biophysical and management factors (variables) that

influence seasonal movements will be identified based on the following lines of evidence:

- a review of available literature and published and digital biophysical data;
- participatory research methods: LEK obtained through discussions in formal and informal meetings with knowledgeable hunters and elders, foresters, biologists, and landowners; and
- a formal workshop on model assumptions with the CBA Disease Risk Management Committee and other partners.

Factors identified as being potentially important in influencing bison movements are:

- habitat type (vegetation and terrain);
- major water bodies;
- agricultural and urban areas;
- linear features (roads, seismic lines, fencing);
- management regime (protection, bison control, open hunting); and
- known presence of bison.

The relative influence of the variables affecting bison movement varies; some are more influential than others. Therefore, variables will be subjectively ranked for relative influence and will be assigned a weighting value. We will use Saaty's (1977) pairwise comparison matrix to develop linear weighted values of importance for each criterion used in the model. In Saaty's procedure, weights are derived by taking the principal eigenvector of a square reciprocal matrix of pairwise comparisons between criteria (Saaty 1977). This can be accomplished through the use of a 9-point continuous scale and a matrix where each variable is compared independently with other variables to determine their relative importance to each other. Each of the model's components is rated relative to the others and assigned 1 of 9 possible "relative importance" values that range from "extremely unimportant" to "extremely important." Weighting values are assigned separately for variables that influence movement across land as opposed to across water or ice. Friction indices ranging from 0 to 10 are assigned to the conditions described by each variable, where a friction index of 1.0 corresponds to no impediment. A value of <1.0 indicates that movement is facilitated, while a value of >1.0 indicates that the variable impedes movement. A friction value is calculated for each pixel comprising the study area. The computation is based on the assigned parameter value modified by the variable weighting. A special condition is applied to pixels that are impassable by bison (e.g., steep cliffs or urban areas).

Once the resistance surface has been constructed, the probability of invasion will be recalculated to reflect local variations in friction.

Model Development and Verification

LEK will be incorporated through a stakeholder involvement process. There are 2 committees currently dealing with the disease issue. The CBA's Disease Risk Management Committee (CBA–DRMC) has participants from the bison and cattle industries, the federal agriculture ministry (Canadian Food Inspection Agency [CFIA]), the Alberta, British Columbia, and Northwest Territories agriculture and wildlife agencies, and 2 First Nations. The Research Advisory Committee (RAC) for WBNP's Bison Disease Research and Containment Program has representatives from the governments of the Northwest Territories and Alberta, 4 First Nations, and non-government environmental organizations. A list of stakeholders and their interests will be developed based on the constituents involved with these committees. A method known as "participatory action research" (PAR; Smith et al. 1997) will be used to incorporate local ecological knowledge and traditional indigenous knowledge into analysis of factors that influence the movement and distribution of potentially infected bison in relation to susceptible ungulate populations. PAR has been developed as an effective technique for researchers to collect epidemiological information at the community level, particularly for identifying and prioritizing information relevant to animal health problems (McCorkle and Mathias-Mundy 1992, Mariner 1998). An application of this approach, termed Participatory Rural Appraisal (PRA), shifts the paradigm from the researcher as an extractor of information, to the researcher as a facilitator of community participation in problem solving (Mariner 1998). This technique was pioneered in the international sustainable development forum as a rapid and economically feasible method for integrating locally available information with scientific data to produce a semi-quantitative product for statistical analysis (Mariner 1998). The flexibility of this technique allows participants to express their interests and contribute information in a qualitative manner that is consistent and meaningful within their conceptual frameworks and knowledge systems. Application of this technique will allow participants to provide information in a visual format on a map surface. Information sought through this process will include the parameters identified for the friction model and others that participants think may influence bison distribution and movements.

Landscape resistance to bison movements will be mapped using a colour scheme or 3-dimensional graphics to represent spatial variations within the areas of interest; areas of greatest risk of invasion will be graphically highlighted. The mapping products, representing integrated scientific and LEK data, will be used to focus discussions among participants on possible alternatives to manage the risk of infection of susceptible populations. Stakeholder involvement will be enabled through the CBA–DRMC and the RAC. Management guidelines will be cooperatively developed for application at the community, industry, and government levels.

MANAGEMENT IMPLICATIONS

The project will provide a focus for clearly defining the interests of participants, including government agencies, the agricultural industry, wildlife and park managers, and First Nations. The methods utilized will provide a forum for bringing these interests together and for integrating scientific and rural appraisal information relevant to the epidemiology of cattle diseases in northern Canada. The models and mapping products produced will serve as a decision-support tool that can be used to:

- assist with rapid evaluations of changes or proposed changes in the distribution and abundance of bison or cattle in the region;
- provide an easily understandable overview of the current or future situations in map form;
- improve the capability of stakeholders to predict the most likely areas where disease could spread so that preventative measures can be planned;
- assist in developing guidelines for management purposes; and
- provide additional insight into the epidemiology of the diseases.

This project will contribute to achieving specific goals and high priority objectives of the Draft National Recovery Plan for Wood Bison, and will contribute to maintaining the disease-negative status of free-ranging bison, commercial bison, and cattle in the region.

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The Success of Ungulate Reintroductions: Effects of Age and Sex Structure

Petr E. Komers[1]
Department of Zoology, Uppsala University
S-752 36 Uppsala, Sweden

G. Peder Curman
Department of Zoology
Uppsala University
S-752 36 Uppsala, Sweden

Björn Birgersson
Department of Zoology
Stockholm University
S-106 91 Stockholm, Sweden

Kenneth Ekvall
Department of Zoology
Stockholm University
S-106 91 Stockholm, Sweden

ABSTRACT

Given that small populations are susceptible to extinction and inbreeding, a primary goal of reintroductions should be to maximize the initial rate of increase. We investigated how the rate of increase of newly reintroduced populations of artiodactyl species was affected by population characteristics, such as size, sex, and age structure. All populations >20 grew positively at a median $r = 0.17$, while several smaller populations declined. Small populations grew faster if they contained more mature individuals. Small, heavily female-biased populations were more variable in r than those of a more equal sex ratio. These female-biased populations also grew on average less well. In sum, populations of <20 were more variable in r than larger ones and both age and sex structure explained a significant portion of this variation. In the experimental part of the study we used fallow deer (*Dama dama*) as a model species. We observed 2 experimental groups: 1 with only socially mature males, and 1 with subadult but physiologically mature males. Females with socially mature males conceived earlier than females with subadult males. The former also conceived earlier than they did in the previous year, while the latter conceived later than in the previous year. (In previous years, the females were exposed to only 1 mature male). Females avoided subadults more than mature males, and they lost more weight than females with mature males. Because subadult males can reproduce, females not mating with them incur 2 costs: energy loss and delayed reproduction. Because females are willing to incur these costs, successful management of small populations requires that a sufficient number of mature males should be present to maximize the probability of fast population growth.

Key words: age and sex structure, conservation, *Dama dama*, fallow deer, population, rate of increase, reintroduction, ungulate.

An increasing number of species are threatened by diminishing population sizes. Small populations are susceptible to extinction due to environmental fluctuations, demographic stochasticity, and inbreeding (Caughley and Sinclair 1994). Reintroduction is a tool used to counteract this threat. To achieve the highest success, reintroductions should maximize the initial rate of increase to shorten the period during which an introduced population would be exposed to these risks. Loss of heterozygocity in particular can be virtually prevented if the founding population is allowed to increase rapidly (Caughley and Sinclair 1994).

Previous studies found that the success of reintroductions increases with habitat quality, and is enhanced if the location is part of the historical range (Griffith et al. 1989, Wolf et al. 1996). Moreover, wild caught animals produce a more successful stock for reintroduction (Griffith et al. 1989). A more recent study confirmed that reintroduction success increased with the number of animals released, but that this success became asymptotic, with the inflection point varying greatly depending on the taxonomic group under investigation (Wolf et al. 1996).

Sex and age composition are absent from current knowledge on the factors affecting reintroduction success, despite their potentially close relationship with population productivity. Anderson (1986) pointed out that the development of a reintroduced population may be temporarily affected by a

[1] Present address: AGRA Earth & Environmental
221-18th Street, S.E., Calgary, AB, T2E 6J5, Canada
pkomers@agrace.com

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

bias in age and/or sex ratio. The age structure of a small population could affect its growth if different aged individuals vary in their reproductive potential. Young, particularly primiparous, mammals are often less fecund than older ones and are more susceptible to environmental stress which could further decrease fecundity (Bronson 1989). Moreover, female estrus is often induced by males (Bronson 1989), and socially mature males could be more efficient at inducing an earlier estrus than younger (but physiologically mature) males (Komers et al. 1999). Hence, socially mature males and females could enhance population growth by producing earlier born, more viable young (Festa-Bianchet 1988, Gaillard and Pontier 1989).

In polygynous mammals in which only dominant males monopolize matings, the risk of inbreeding is obvious. While female choice or the outcome of male competition can hardly be managed in wild populations, the structure of a reintroduced population can be manipulated. Typically, this is done to maximize the number of females and hence the potential number of young produced. The result is expected to be a rapid population increase, counteracting the loss of heterozygocity. However, a low density of males may not effectively induce estrus, resulting in either delayed or even failed reproduction (Walkden-Brown et al. 1993).

There is unequivocal evidence that male presence can induce ovulation in many mammalian species (Bronson 1989). However, changing the emphasis to say that females detect the adequate stimulus would mean that the female can control her own estrus. Female control of estrus is the foundation of arguments that females either synchronize, advance, or postpone their estrus adaptively (Knowlton 1979, McClintock 1983, Pereira 1991). It also would be an essential tool for females in expressing their mating preferences.

If females adjust estrus to the availability of preferred males, then they should ovulate when desired males are available and delay ovulation when only undesired males are available. Delaying ovulation in the presence of reproductively active males, however, has at least 2 costs: delayed or forgone production of young and the energy cost of avoiding the advances of these males. Both costs would have serious implications on the productivity of a small population where the number of males of a certain age class would be limited.

To address growth patterns in reintroduced populations in a review of reintroductions, we investigated how population characteristics, such as population size, sex, and age structure, affected the rate of population increase. We focused on Artiodactyla because these species attract much attention in conservation due to their potential role as flagship species, game or harvest species, and as charismatic species attracting great public attention, and due to the high costs in their translocations (Boyer and Brown 1988). To address the question of how such patterns of population growth are produced, we established 2 experimental populations using fallow deer (*Dama dama*) as a model species.

METHODS

Review of Past Reintroductions

We obtained most data for this review from published literature (Table 1) and personal communication with people involved in reintroduction projects with incompletely published records. We excluded potentially confounding variables by following the recommendations of Griffith et al. (1989). Accordingly, we analyzed only populations that originated from a wild-caught stock and that were introduced to the historic range of the species. By focusing on Artiodactyla we excluded confounding variables stemming from differences in life histories. Although the absence of potential competitors seems to improve the reintroduction success (Griffith et al. 1989), we included reintroduction only in locations where another ungulate species existed because: (1) most ecosystems include >1 ungulate species; and (2) the vast majority of translocations that we investigated were performed under these conditions. We did not consider factors that were not associated with the success of translocation of mammals such as period of acclimatization, soft or hard (no food provided on release site) release, the number of releases, and other factors investigated in previous reports (Griffith et al. 1989, Wolf et al. 1996).

The sex structure was calculated as the male/female ratio and the age structure as the proportion of socially mature animals. Where the growth rate, r, was not calculated in the published report, we used the formula $r = [\ln(N_t/N_0)]/t$, where N_t is the number of animals alive t years after reintroduction, N_0 is the number of animals released, and t is the time after reintroduction in years. In those reports in which releases had been conducted in consecutive years but data about births, deaths, and new releases were given (Krasinski 1978, Krasinski and Krasinska 1992, Dunham *et al.* 1993), we calculated the rate of increase for each of the years with sufficient data. Then, we used the mean of these values in the analysis. In 1 publication, the authors themselves provided a value of rate of increase (Gogan and Barret 1988). We included only populations for which r could be calculated over 10 years after reintroduction. This arbitrary limit is based on the argument that within this time even small populations can grow to a size at which demographic effects become negligible (Caughley and Sinclair 1994).

In the analyses of age and sex structure, we excluded populations <5 because of the uncertain interpretation of whether the respective ratios represent a measure of the balance or simply small numbers. For example, the sex ratio in a pair can only be 1.0, but the female has access to only 1 male. All data sets were checked for normality. In regression analyses, r was transformed to $r' = (r+1)^4$. Stepwise multiple regression was applied to identify those predictor variables

Table 1. Introduced artiodactyl populations and their characteristics.

Species	No. of animals	Ages(socially mature/total)	Sex ratio (male/female)	Rate of increase
Mouflon[a] (*Ovis musimon*)	2	—	1.00	-0.173
Mountain goat[b] (*Oreamnos americanus*)	2	1.00	1.00	0.090
Mouflon[c]	3	—	0.50	0.245
Mouflon[c]	4	—	1.00	-0.231
Mountain goat[d]	4	—	—	-0.116
Mouflon[c]	5	—	0.25	-0.536
Muskox[e] (*Ovibos moschatus*)	5	0.40	0.67	0.166
Mouflon[c]	5	—	0.67	-0.268
Tule elk[f] (*Cervus elaphus nannodes*)	7	1.00	1.33	0.290
Mouflon[c]	7	—	0.40	0.206
Mouflon[c]	8	—	1.67	0.305
Mountain goat[g]	8	—	0.33	-0.200
Mouflon[c]	9	—	0.80	0.172
Tule elk[f]	10	1.00	0.25	0.200
Mountain goat[h]	11	0.54	0.83	0.100
Mouflon[c]	13	—	0.30	0.204
Wood bison[i] (*Bison bison athabascae*)	13	0.40	—	0.217
Mouflon[c]	14	—	0.75	0.141
Wood bison[i]	14	—	—	0.100
European bison[j] (*Bison bonasus*)	14	0.21	1.00	0.109
European bison[k]	15	0.13	0.50	0.030
Mouflon[c]	17	—	0.89	0.122
Wood bison[i]	18	0.55	0.40	0.215
Mouflon[c]	20	—	—	0.268
European bison[j]	21	0.19	0.88	0.151
Mountain goat[g]	21	—	0.75	0.065
Bighorn sheep[d] (*Ovis canadensis*)	21	—	1.10	0.178
Reindeer[l] (*Rangifer tarandus*)	29	0.50	0.21	0.322
Bighorn sheep[d]	31	—	—	0.069
Bighorn sheep[b]	31	0.93	0.26	0.110
Bighorn sheep[m]	41	0.49	0.18	0.200
Mountain gazelle[n] (*Gazella gazella*)	54	—	0.50	0.197
Bighorn sheep[m]	69	0.55	0.27	0.110

[a]Niethammer 1963; [b]Oldenburg 1996, M. Scott, Id. Dep. Fish and Game, pers. comm.; [c]Magnani 1978; [d]J. B. Stelfox, Forem Consulting, pers. comm.; [e]Lundh 1979; [f]Gogan and Barret 1988, Mendelssohn 1993; [g]Matthews and Coggins 1994; [h]J. Jorgenson and R. Quinlan, Alta. Environ. Prot., pers. comm.; [i]C. C. Gates, Northwest Territ. Dep. Renewable Resour., pers. comm.; [j]Krasinski and Krasinska 1992; [k]Krasinski 1978; [l]Klein 1959, 1968; [m]Cook et al. 1990; [n]Dunham et al. 1993.

that explain most of the variance while accounting for the remaining variance explained by others. The specific tests applied are identified in the text. All populations in Table 1 were entered in the analysis as individual data points in a pool, irrespective of the number of times that a species was represented. This approach was taken because aggregation of data by species reduced the variance of the pooled data set by about 3-fold (due to the high intraspecies variation). In such a situation pooling does not bias results (Leger and Didrichsons 1994).

Experimental Populations

We conducted our experiments at the Tovetorp research station of Stockholm University where captive fallow deer were under investigation from 1991 to present. In April 1996, we established 2 experimental groups which we observed during the rut in the following autumn. They were separated by about 1 km of forest and meadows. The vegetation in the enclosures was mainly grassland with single coniferous and deciduous trees. The 2 enclosures were similar in food availability, tree cover, and number and reproductive history of animals: 15 sexually experienced females (>2.5 years old, mean date of conception was 4 November 1995 in both groups); 6 sexually inexperienced females (1.5 years old); 7 fawns; 6 spikes (1.5-year-old males); and 6 sexually mature males. The areas of the enclosures were 2.9 and 3.5 ha, respectively. However, the groups differed in the age of adult males: in the young-male experimental group (hereafter

YM), all adult males were 2.5 years, while all were 5.5 years in the mature-male group (hereafter MM). All males were sexually inexperienced. In YM, 1 female and her fawn escaped into a large adjacent enclosure (void of other deer) so that there were 14 mature females. All animals were marked with unique colour tags. The animals were weighed with a cattle scale placed in front of the feeder where the deer came to feed in the evenings and nights. Food was provided only after daylight observations and the supply was depleted before the next morning.

The 2 experimental groups were monitored simultaneously from 15 October to 12 November by 6 observers divided into 2 teams. To avoid observer bias, the 2 teams rotated between experimental groups at 1200 every day. Observations were performed during all daylight hours. Because group sizes were small and all animals could be seen at anytime, we recorded all interactions using the "all occurrence" sampling method (Altmann 1974). For each interaction, we recorded identities of the initiating and receiving individual. The following interactions between males and females were recorded: *male advance* (male approaches a female to within 1 body length with his head turned sideways and his cheek facing the female; this appears to be a courtship behaviour), *move away* (female moves away following a male advance), *run* (female runs, usually with the male in pursuit). We also recorded all copulations. From dusk until 2400, 1 observer recorded copulations only. Observations resumed at 0500. To enable identification of the animals and observation of copulations after dusk, the scale and the nearest surroundings were illuminated. We observed 19 copulations and inferred from behavioural data 3 and 2 additional copulations in YM and MM, respectively. These copulations presumably occurred after 2400. All 19 observed copulations were preceded during 1 day by high levels of interactions between the respective female and the males. The inferred copulations indicated similarly elevated levels of interactions during one day. Births in the subsequent spring supported our inference.

All data sets were tested for deviations from normal distribution. Percentages were analyzed after applying the arcsine transformation. Nonparametric tests were applied on data sets which were not normally distributed. Throughout this section of the paper, sampling units were the means calculated (or dates observed) for each individual animal. Hence, sample sizes were the numbers of animals in each group, age, or sex class (see previous description of experimental groups).

RESULTS

Overall Relationships

Our results corroborate the conclusions of Wolf et al. (1996) that reintroduction success increases with the number of animals released (Fig. 1). In a refinement of this conclusion, we found that the increased success of larger artiodactyl populations was due to their higher rate of increase. From Fig. 1 we conclude that populations of ≥20 animals showed exclusively a positive growth (median $r = 0.17$). Among smaller populations, the median was $r = 0.12$, and 6 of 23 showed negative growth. Moreover, the variance in residuals around the trend line in Figure 1 was greater in populations <20 than in larger ones (2-sample F-test for variances, $F_{22,9} = 4.3$, $P = 0.015$). This result suggests that the growth rate of small populations was affected relatively more by factors other than population size.

Responses of Populations <20

Given the exclusively positive growth of populations ≥20, we focused on population characteristics that affect the rate of increase of smaller populations. Among these, only the age structure explained a significant portion of the variance while the number of animals released and the sex ratio did not affect the rate of increase significantly (Fig. 2; stepwise multiple regression, $F = 10.4$, df = 6, $R^2 = 0.68$, $P = 0.023$). This result demonstrates that population growth increased with the proportion of mature animals in the population.

The power of this analysis was limited by the low sample size ($n = 7$) of populations with >5 animals which include information on all 4 variables (Table 1). To increase the power

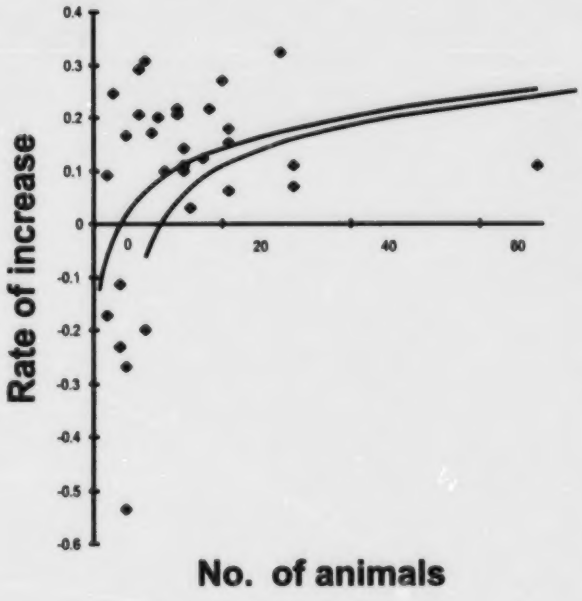

Figure 1. Rate of population increase in relation to number of animals in reintroduced ungulate populations. The trend line was calculated as $y = 0.09 * \ln(x) - 0.15$, $R^2 = 0.17$.

of the analysis, we reanalyzed the data for 16 populations >5 animals with data on sex ratios using only the number of animals released and the sex ratio as predictor variables for r. This second analysis was justified because the proportion of mature animals was not correlated with sex ratio (Pearson correlation coefficient = 0.09, n = 7, P = 0.85) and hence could not confound the result. Growth rate r increased with the proportion of males in the population (Fig. 3; stepwise multiple regression, F = 5.05, df = 15, R^2 = 0.27, P = 0.041). However, this positive relationship was mostly based on the larger variance in populations with few males, in which some populations grew poorly while others grew well: populations with a sex ratio <0.67 (the median sex ratio of reintroduced populations) had a higher variance of r than populations with a higher sex ratio (Fig. 3; $F_{7,7}$ = 11.3, P = 0.004).

Our findings are relevant for a great majority of recent translocations, and are further exemplified by records we obtained from the State of Idaho (these records include all translocations, irrespective of origin, release site, and calculations of r). Since 1985, 78% (46 of 59) of all translocations were conducted with populations of <20 sheep or goats (Oldenburg 1996; J.J. Beecham, Id. Dep. Fish and Game, pers. comm.). Moreover, 91% (31 of 34) involved sex ratios of <0.67 .

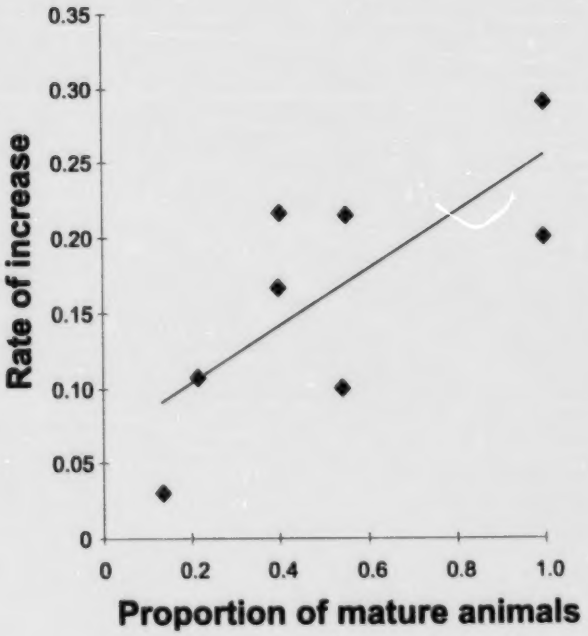

Figure 2. Rate of population increase in reintroduced ungulate populations in relation to the proportion of socially mature animals in populations of <20 animals. The trend line was calculated as $y = 0.20^{*}x + 0.07$, $R^2 = 0.62$.

Experimental Populations

Between 1991 and 1995, when all females at the research station were exposed to only 1 mature male, the mean date of copulation was 2 November (range: 31 October–4 November). In 1996, when females were exposed to 6 mature males in MM and 6 young males in YM, the mean date of copulation was earlier in MM and later in YM than in all previous years (5 of 5 years were later or earlier, respectively; binomial, P = 0.06). Females mated 9 days later in YM than in MM (mean dates: 28 October and 6 November, in MM and YM, respectively; t' = 6.99, P < 0.001; Fig. 4).

All observed copulations in both experimental groups were followed by birth after 231 days (YM: ± 2.9 days, n = 12; MM: ± 3.1 days, n = 12; 1 copulation in each group was not detected but was indicated by birth the following spring). This result conforms to the gestation length recorded in the research station in previous years (Birgersson and Ekvall 1997), and indicates that all copulations coincided with ovulation. Only 2 females were observed to mate twice. In both cases the second copulation occurred on the same day as the first. In 1997, females in YM gave birth 3.1 ± 3.8 days later than in the previous year, while MM females gave birth 6.9 ± 6.7 days earlier (paired t-tests; YM: t = 2.2, df = 12, P = 0.014; MM: t = 2.2, df = 11, P = 0.004). These values indicate that females in both groups changed their timing of conception significantly relative to the breeding season before our experiment. Besides the mean date of copulation, the difference in the variances of copulation dates was highly significant ($F'_{12,11}$ = 5.34, P = 0.009), the females in YM being more synchronized (Fig. 4). Sexually inexperienced females

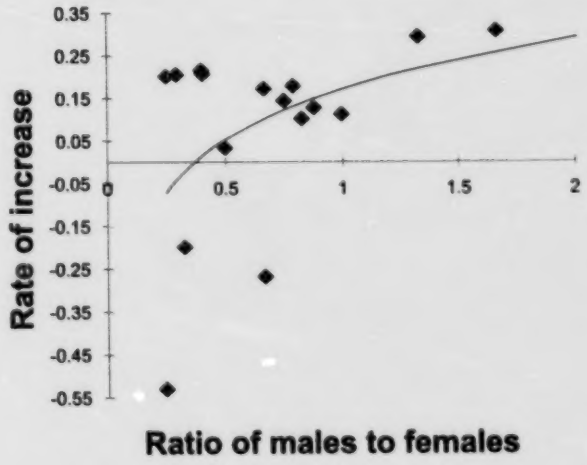

Figure 3. The rate of population increase in reintroduced ungulate populations in relation to the ratio of males to females in populations of <20 animals. The trend line was calculated as $y = 0.17^{*}\ln(x) + 0.17$, $R^2 = 0.20$.

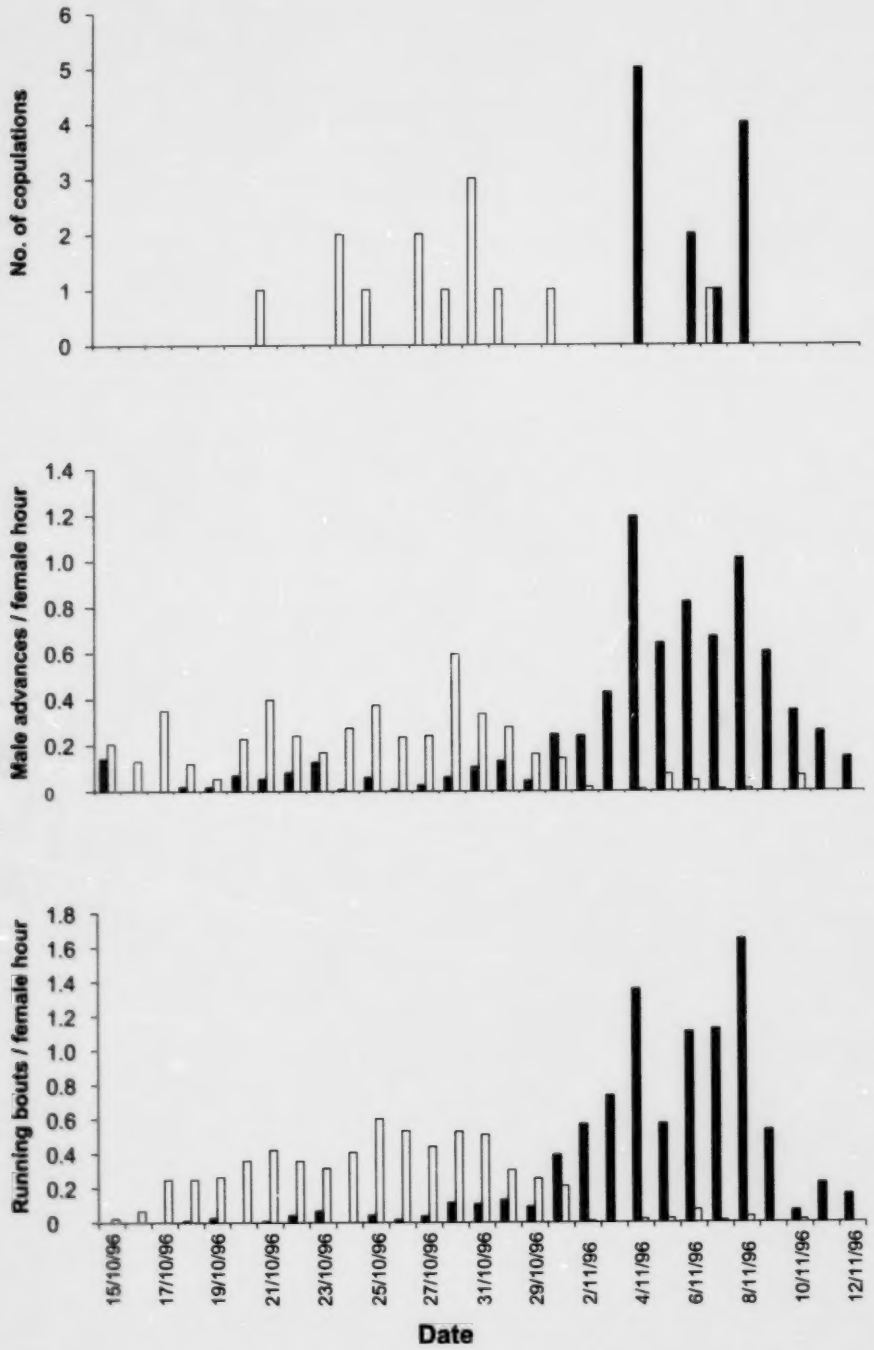

Figure 4. Top: Time distribution of copulations in 2 experimental groups of fallow deer in Sweden. Centre: Chronology of male advances towards fallow deer females, expressed as advances per female and hour. Bottom: Chronology of fallow deer female running bouts, expressed as bouts per female and hour. Open bars represent the experimental group with socially mature males. Black bars represent the group with only subadult males.

were not observed to copulate in either enclosure.

If females prefer mature over young males, they should avoid the advances by young males. Accordingly, YM females moved away in 55% (± 1%) of male advances while MM females moved away in 46% (± 2%, t = 3.4, P = 0.002). YM males also advanced more often towards females (Fig. 4; 0.26 ± 0.02 male approaches/hr in YM versus 0.18 ± 0.02 in MM, t = 2.72, df = 27, P = 0.011) and YM females ran more often than MM females (Fig. 4; 0.32 ± 0.01 running bouts/hr in YM versus 0.23 ± 0.02 in MM, t = 2.77, df = 27, P = 0.010). Moreover, females in YM lost 4.2% (2.4 ± 0.4 kg) of total body weight during the rut, while in MM females lost 1.9% (1.1 ± 0.3 kg; repeated measures ANOVA, interaction of measurement–time by group, F = 6.81, df = 1, P = 0.016).

DISCUSSION

Effects of Population Size

We found a similar relationship between the number of animals released and the rate of increase as reported for a wider range of mammals (Griffith et al. 1989, Wolf et al. 1996), namely that reintroduction success initially increases substantially with the size of the founder population but that the function becomes asymptotic at about 20 animals. Moreover, our results show that the variation in the rate of increase of reintroduced artiodactyl species was larger when <20 animals were released. This result was expected, because demographic and environmental stochasticity affect smaller populations more than larger ones (Caughley and Sinclair 1994). Our results show that a significant portion of the variation in small reintroduced artiodactyl populations was explained by their age and sex structure. In particular, the proportion of socially mature animals and the proportion of males were positively related to population growth.

Age-Structure Effects

The age of introduced animals in populations <20 appeared to be important for the success of reintroductions because populations with more mature animals grew faster. This finding could be explained by a higher fecundity of mature females (Bronson 1989). Furthermore, socially immature males of several ungulates were found to harass females more than mature males (Shackelton 1991, Clutton-Brock et al. 1992, Komers et al. 1994). This behaviour could reduce female body condition (Komers et al. 1999).

Male mammals often induce the estrus of females (Bronson 1989). Mature males of some ungulate species seem to induce estrus more effectively than young males and they tend to be preferred as mating partners by females (Miquelle 1991, Clutton-Brock et al. 1992, Komers et al. 1999). We infer that if <20 animals are available for a reintroduction, socially mature animals should predominate.

Sex Ratio Effects

Populations with a female-biased sex ratio had a wide range of rates of increase. In populations with a high female bias mature males were rare and may not have impregnated all females within the normal rutting period. This could be particularly true for small populations in which individuals may be relatively dispersed. Additionally or alternatively, a low exposure to males could have negative effects on the fecundity of females (Walkden-Brown et al. 1993, Komers et al. 1999).

The positive relationship between the proportion of males and population growth was largely based on the increased incidence of poorly growing populations in which males were rare. Some of these populations, however, still grew well resulting in a large range of growth rates in heavily female biased populations. Intuitively, population growth is expected to decrease again once the sex ratio surpasses 1.0. However, we have found no reintroduced populations that were strongly biased towards males.

Experimental Populations

Female fallow deer in the experimental study ovulated earlier when exposed to socially mature males (in the MM experimental group), than when only young males were available (in the YM experimental group). The finding that each observed copulation resulted in a young being born after a normal gestation length for fallow deer (Birgersson and Ekvall 1997) indicates that copulations coincided with ovulation. In YM, this ovulation is unlikely to have been preceded by a silent estrus (no behavioural signs of physiological estrus) because the estrous cycle of fallow deer is 22 days (Asher 1985). This finding would imply that the silent estrus in YM would have to have begun on average 13 October, 15 days earlier than the average copulation in MM where the timing of the rut coincided with the rut of wild fallow deer in Sweden (B. Birgersson, K. Ekvall, and P. E. Komers, unpubl. observations). Therefore, the delay of estrus in YM is due to a change in the timing of estrus and not simply due to the omission of estrous behaviour.

Our findings show that females avoid young males at the cost of losing more body mass. This behaviour could affect their survival because female ungulates reducing their body reserves before the winter may compromise their own winter survival (Robbins 1983, Leader-Williams 1988). Moreover, delaying the birth date of young results in decreasing their viability because late-born young may enter the winter in poorer body condition. Reduced viability is found in ungulates in which late-born fawns enter the winter with a lower body mass and show a higher winter mortality (Festa-Bianchet 1988, Gaillard and Pontier 1989, Birgersson and Ekvall 1997).

In sum, females in YM suffered at least 2 costs: delayed production of young and increased loss of body mass, despite

the ability of young males to sire offspring. This ability to sire was demonstrated by the courtship behaviour of young males such as their advances to females and by each copulation resulting in a young (Komers et al. 1997).

The alternative argument that young males are simply less potent stimulators implies that the stimulus would be below a threshold which would be required for females to come into estrus. Since intensity of stimulus is positively related with synchrony of estrus in females (McClintock 1983, Bronson 1989, Walkden-Brown et al. 1993) the argument finds little support in our study because under a lack of stimulus, females in YM should be less, not more, synchronized in estrus.

Female mammals often affect each other in their estrous cycle and the effect could be either one of enhancement or of suppression (McClintock 1983, Bronson 1989). Conceivably, only 1 female in MM responded to the stimulus of the mature males, and her estrus, not the presence of the males, advanced the estrus of the other females. Although this would effectively reduce our sample to one in this group, the result is still relevant on a population level indicating that a group of gregarious mammalian females advance their estrus when preferred males are present. In YM, however, females delayed their estrus relative to previous years. Although female mammals are known to suppress the estrus of others in certain situations, all studies show that for suppression to occur ≥1 female must be in estrus or in impending estrus (McClintock 1983). Moreover, synchronization of estrus (to the extent observed in YM) by suppression is usually >1 estrous cycle (McClintock 1983). Because we found no evidence for estrous cycles in YM before the copulations, females in YM apparently could not have suppressed each other in the timing of estrus.

CONCLUSIONS

By limiting our analysis to those reintroductions which followed the procedure that has already been determined as promoting success (Griffith et al. 1989, Wolf et al. 1996), we put ourselves in the situation of a manager who attempts to maximize success applying current knowledge. Our results support the current recommendations for successful reintroductions because populations ≥20 grew well, while variation in growth was high among smaller ones (Griffith et al. 1989, Wolf et al. 1996). Following the arguments of Griffith et al. (1989), augmenting the number of animals beyond 20 does not apparently increase the chances of success, but dividing a population of 40 in 2 populations of 20 would increase the chances of long-term success when environmental catastrophes could negate the initial population growth. Our results suggest that success of reintroductions of populations <20 could be enhanced by increasing the proportion of mature animals and that a sex ratio close to 1.0 would minimize variation by reducing the number of poorly growing founder populations.

This conclusion is supported by our study in which female fallow deer adjusted their date of estrus to the age of reproductively active males. Because females appear to be willing to delay reproduction at the cost of decreased body weight and later birth date of their young, we conclude that populations in which females have limited access to mature males may suffer decreased probability to grow at high rates.

We have only investigated artiodactyl species. Our results may be representative for other species of mammals. However, because our arguments on age-fecundity relationships and a polygynous mating system seem to be valid for most mammalian species (about 95% are polygynous) (1977), the trends of our results may hold for many mammals. Actual population sizes at which reintroductions may be considered safe, however, will probably vary with species specific characteristics such as intrinsic rates of increase (Caughley and Krebs 1983).

ACKNOWLEDGEMENTS

For valuable discussion and comments, we thank R. D. Brown, J. Byers, M. Festa-Bianchet, J. Höglund, O. Leimar, K. McComb, T. Radesäter, J. B. Stelfox, I. Stevenson, H. Tamrin, S. Ulfstrand, C. Wiklund, and an anonymous reviewer. Names and addresses of people involved with reintroduction projects were provided by P. S. Soorae at IUCN/SSC Reintroduction Specialist Group in Nairobi, and by J. B. Stelfox at Forem Consulting. We thank N. Andbjer, A. Bylin, P. Curman, C. Pélabon, and S. Strömberg for valuable discussion and enthusiastic field assistance. D. Stenström provided recordings of groaning fallow deer and I. Stevenson helped with computing. Financial support was provided by the Swiss Academy of Sciences, the Roche Research Foundation, the Ciba-Geigy Jubiläumsstiftung, the Swedish Natural Science Research Council, the Foundation Lars Hiertas Minne, and the Royal Swedish Academy of Sciences.

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Relative Importance of Episodic Versus Chronic Mortality in the Decline of Vancouver Island Marmots

Andrew A. Bryant
Andrew A. Bryant Services
108 Fifth Street, Nanaimo, BC V9R 1N2
abryant@island.net

ABSTRACT

I used population count data collected from 1979 to 1998 to evaluate the relative importance of "chronic" versus "episodic" mortality patterns in Vancouver Island marmots (*Marmota vancouverensis*). I hypothesized that some factors (e.g., gradual vegetation change, forestry effects, or predation) should produce chronic patterns of mortality, and others (disease and weather) should produce episodic patterns. A few colonies had consistently low survival. These results are consistent with a hypothesis of poor quality "sink" habitats that are maintained by immigration from nearby "sources." However, most colonies had distinct episodes of high mortality, and these appear to be the primary cause of recent population declines. Spatial and temporal patterns of survival are consistent with a hypothesis of disease outbreak or increased hunting effort by predators within a small geographic area. The impact of both factors was probably exacerbated by forestry activities. Results underscore the precarious status of this endangered mammal and illustrate the value of carefully designed annual population surveys.

Key words: *Marmota vancouverensis*, mortality, population surveys, Vancouver Island marmot.

Caughley and Gunn (1996) offered a straightforward model for managing endangered species. First, determine whether populations are declining or whether other evidence suggests that a problem exists. Second, learn about the ecology of the organism and use the accumulated knowledge to construct hypotheses about possible causal factors. Third, subject the resulting hypotheses to rigorous scientific testing. Finally, use the results to reverse the factors that are causing the problem. In this paper I use long-term population count data from a critically endangered mammal to address a basic question: can annual population count data be used to test hypotheses about the causes of decline?

Vancouver Island marmots (*Marmota vancouverensis*) are endemic to Vancouver Island, B.C. (Nagorsen 1987). Apart from small colonies on Mount Washington, all known active sites are located within 5 adjacent watersheds on southern Vancouver Island (Bryant and Janz 1996). Recent population trends on southern Vancouver Island are intriguing. Many colonies expanded during the early 1980s and this was accompanied by colonization of some new habitats created by clearcut logging of forests above 700 m elevation. Unfortunately this expansion was limited both in geographic and temporal terms. Despite successful reproduction and survival in new habitats created by forest harvesting, marmot numbers subsequently declined from a peak of more than 300 animals during the mid-1980s to the present total of fewer than 100 animals (Bryant 1998).

Several hypotheses have been advanced to explain recent population dynamics, including natural habitat change (Milko 1984), source-sink and landscape connectivity dynamics caused by forestry (Bryant 1996), periods of inclement weather (Smith 1982), changing predator abundance (Merilees 1980), and disease outbreaks (Bryant et al. in press). These hypotheses are not mutually exclusive and there is no a priori reason to imagine that a single factor is responsible for observed population trends. However, the hypotheses are useful in generating testable predictions and exploring the fundamental question of whether declines are primarily due to what I describe as "chronic" or "episodic" processes.

CHRONIC PROCESSES

A variety of mechanisms could reduce habitat quality over time. In natural subalpine meadows, possible mechanisms of habitat change include invasion by trees or bracken ferns (*Pteridium* spp.), post-fire succession, and changing foodplant availability (Milko 1984, Martell and Milko 1986, Laroque 1998). Gradual processes such as these lead to the expectation of slow decline in survival or birth rates as individual habitats become increasingly unsuitable. There is no reason to expect that change would occur simultaneously at

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

all sites or that there would be an abrupt temporal threshold between habitats being "suitable" or "unsuitable." Given the short duration of this study compared to rates of change in subalpine meadows (e.g., Schreiner and Burger 1994), one would expect to observe chronic low birth or survival rates at a subset of natural colonies.

In clearcuts, forest regeneration is the principal mechanism of change, and this process is exceptional because it can act over a temporal scale measured within the lifetime of individual marmots. The successional state of regenerating clearcuts could influence birth or survival in a linear fashion (i.e., a gradual reduction as a function of increasing forest age). Alternatively, it could be manifested by a threshold effect, in which conditions become unsuitable for birth or survival over a period of a few years. In either case the prediction is that when averaged across years, survival should be relatively low in clearcut habitats.

It is likely that mortality due to predation would be reasonably consistent across years and colonies. Known marmot predators such as wolves (*Canis lupus*), cougars (*Puma concolor*), and golden eagles (*Aquila chrysaetos*) are relatively long-lived compared to marmots, and their populations should not fluctuate greatly from year to year. In addition, individual predators are likely to benefit from learned hunting behaviour, and individuals would be expected to continue hunting at locations where they were successful in the past. For these reasons the predation hypothesis would be expected to produce chronic mortality patterns.

EPISODIC PROCESSES

Several factors could cause episodes of high mortality and therefore produce abrupt shocks to the marmot population. For example, marmots might suffer from "bad weather" years resulting from extremes in snowpack or rainfall (e.g., Armitage 1994). Disease events (epizootics) could also occur. In this case the fundamental prediction is that marmots would display "episodes" of high mortality. Weather effects would be expected to occur randomly over time, but disease events might occur over a period of a few consecutive years as the infection spreads.

SPATIAL CORRELATION OF SURVIVAL

Given the small size of the geographic area occupied by marmots, one would expect weather to influence the entire metapopulation. There should, therefore, be no spatial correlation of survival as a function of between-colony distance (i.e., there should be uniform correlation throughout the study area). Patterns of vegetative change in natural meadows or in clearcuts should also be independent of spatial location within the landscape.

In contrast, the disease hypothesis suggests that survival would be more highly correlated at nearby colonies due to increased risk of infection (May and Anderson 1979). Similarly, because predators may focus hunting efforts at locations that offer the highest probability of success, one would also expect mortality to be spatially correlated.

STUDY OBJECTIVES

My primary objective was to test whether spatial or temporal patterns of survival were congruent with any or all of the above hypotheses. My secondary objective was to illustrate the utility of some basic analytical tools in evaluating hypotheses about the causes of decline.

STUDY AREAS

The present study encompassed the known marmot metapopulation on southern Vancouver Island. Data from Mount Washington were excluded for reasons of small sample sizes, because colonies were infrequently sampled, and because it is unlikely that dispersal occurs between that mountain and the southern metapopulation (Bryant and Janz 1996). Within this area of approximately 150 km2 all colonies were mapped (1:50,000 National Topographic Series) and entered into a Geographic Information System (Arc/Info) to facilitate spatial measurements.

METHODS

Population Counts

Marmot counts were made by many people (see Acknowledgements). Most counts were conducted before 1100 to coincide with known marmot activity rhythms. Counts were usually repeated 2–4 times annually, although some sites were visited only once, and not every site was visited in each year (Bryant and Janz 1996).

Compilation and Survival Estimation

I used a simple measure of marmot survival for all analyses. This measure was calculated from consecutive annual counts in which apparent survival, ϕ, was the count of adults at time $t+1$ divided by the count of adults and pups at time t. Using Pulliam's (1988) terminology, this actually represents "*i-d-e*" or net immigration–death–emigration, and will therefore accurately reflect survival only if immigration equals emigration. Mark–recapture work (Bryant 1998) shows many cases in which this assumption is violated, but because I was interested in the relative importance of episodes of high mortality I reasoned that slight changes in annual immigration:emigration rates would not mask overall trends. For the same reason, I assumed the upper boundary of survival to be 1.0 despite numerous instances in which colonies

increased because of immigration. For example, the number of non-reproductive adults at Butler Peak "west roads" colony increased from 2 in 1982 to 9 in 1983. In this case I assumed survival to be 1.0 and not 4.5.

Finally, the shape of the population count data dictated that I focus analyses on changes between the "early period" (1981–87) and "late period" (1991–97). This was necessary because in the intervening years there was no inventory apart from the few sites that academic researchers or naturalists visited. Even in the high sampling years, marmot counts suffered from inconsistent sampling effort.

Statistical Analysis

I used locally weighted regression (LOWESS; Cleveland 1979) to elucidate trends in adult abundance within colonies and across time. This method has the effect of minimizing variance caused by "pulses" of reproduction (female marmots don't breed every year and few colonies contain more than a single reproductive-age female) or differences in sampling effort across years (see above). It therefore provides a useful overview of whether all colonies declined, and I used Pearson correlations to test for this (Zar 1974). However, by definition, employing "smoothers" such as LOWESS remove the peaks and lows of survival rates, therefore removing the very episodes of high mortality that are the subject of this paper.

Autocorrelation describes the pattern in which nearby things are more or less alike. In this case the word "nearby" can be interpreted in both a geographic (spatial) context and a temporal (year-to-year) context. To address the question of temporal autocorrelation I calculated coefficients of variation (CV) to evaluate the magnitude of year-to-year fluctuation in survival rates (Zar 1974). Colonies with chronic mortality patterns should show relatively low coefficients of variation.

To evaluate spatial autocorrelation I used Moran's I coefficient. The general form of the statistic is analogous to Pearson's R in that it is a general cross-product statistic (Sokal and Oden 1978). Values close to 1.0 indicate that similar values tend to cluster together in a spatial context, and values approaching -1.0 indicate that dissimilar values tend to cluster together. The essential formula is:

$$I = \Sigma\Sigma W_{ij}\Delta_{ij}$$

in which Δ_{ij} is a measure of the proximity of the variate (in this case apparent survival) between the $_i$th and $_j$ th spatial positions, and W_{ij} is a spatial weighting function. A fundamental requirement of spatial correlation analysis is some measure of contiguity or "connectivity" of the locations. On a regular grid (e.g., quadrat data), contiguity (W_{ij}) is typically set to equal 1 for nearest neighbours and 0 if otherwise (Haining 1990). For

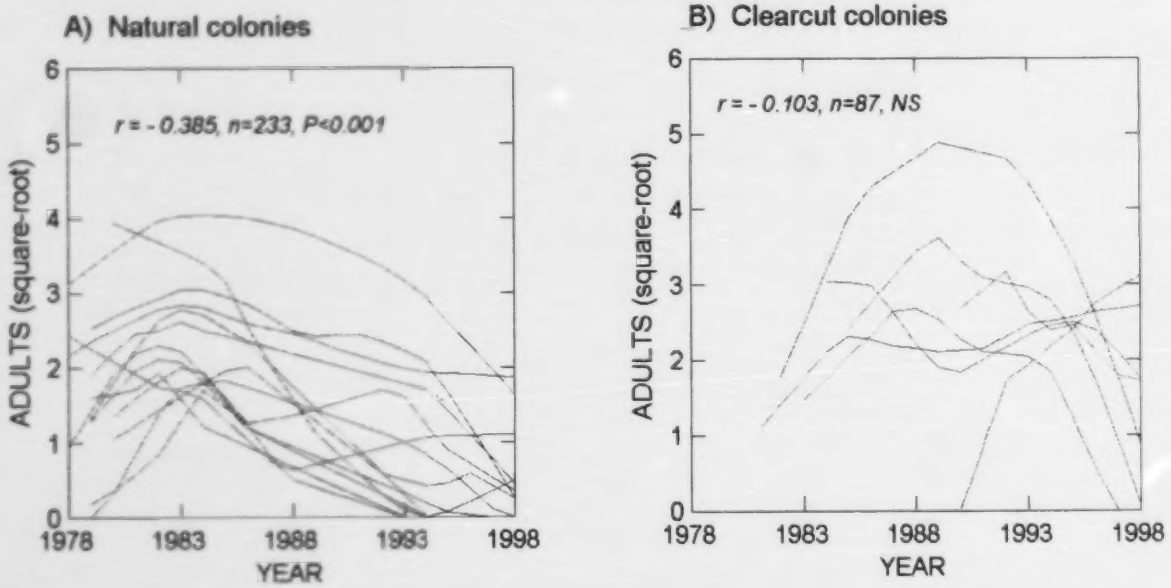

Figure 1. Population trends within and among colonies. Data are LOWESS regressions of maximum annual numbers of adults at natural meadows (A) and clearcuts (B). Abundance data were square-root transformed to facilitate comparison of trends among colonies of different sizes. Marmots in natural colonies declined significantly over time (Pearson correlations) but increased during the early 1980s. Marmots in clearcuts increased later, but most colonies subsequently declined.

Table 1. Variation in colony-specific survival. Data are median survival rates for 16 Vancouver Island marmot colonies with at least 5 years of data. Several additional sites (e.g., Mount Whymper, "P" Mountain, and Mount Buttle) showed similar trends, but sporadic sampling effort precluded analysis. (N = population size, CV = coefficient of variation.)

		1981–87			1991–97		
Colony	Type	N	Median	CV	N	Median	CV
Butler Peak	clearcut	4	0.93	13	6	0.43	57
Bell Creek	natural	7	0.90	21	7	0.50	72
Vaughan Road	clearcut	5	0.90	34	5	0.50	83
Gemini	natural	6	0.75	50	*extinct*		
South Green	natural	5	0.75	42	*extinct*		
Haley Lake	natural	7	0.70	29	7	0.50	37
Westerholm	natural	6	0.68	60	*extinct*		
K44E	clearcut	7	0.67	33	7	0.75	46
D13E	clearcut	4	0.66	35	5	0.75	30
Green Mountain	natural	7	0.65	29	7	0.80	39
G2C	clearcut	5	0.60	44	*extinct*		
West Green	natural	6	0.58	62	*extinct*		
Green NW	natural	6	0.51	74	*extinct*		
Pat Lake	clearcut	1			6	0.63	29
Sherk Lake	clearcut		*not yet colonized*		6	0.89	24
Mount Franklin	clearcut		*not yet colonized*		6	0.61	14

more spatially complex data (such as locations of marmot colonies), contiguity can be calculated using the principle of an "effects radius" (Smith and Gilpin 1997).

For marmots, I reasoned that mortality due to weather would occur over the entirety of the Nanaimo Lakes metapopulation, but that mortality due to predators or disease would occur over progressively smaller areas. I therefore tested for spatial autocorrelation using a cumulative effects radius in which only those neighbours within the radius were tallied as being contiguous (i.e., $W_{ij} = 1$). Significance of Moran's I was evaluated using Z scores (Sokal and Oden 1978). The null hypothesis in this case is that the observed distribution of events is no different from a distribution in which values are randomly assigned to the same set of spatial locations.

RESULTS

Population Trends among Colonies

Sequential population estimates could not be made for many colonies because of gaps in sampling coverage, but LOWESS regressions were useful in exploring trends over time. Apparently no colony remained stable over the study period (Fig. 1). Marmot numbers in natural habitats declined systematically with year. The data suggest a definite upward bulge that occurred in 1983, which probably reflects high reproduction in 1982 or earlier. Numbers of marmots in clearcuts increased greatly during the late 1980s (the Butler Peak "west roads" colony was the largest), but from about 1992 on most colonies declined rapidly.

Chronic and Episodic Patterns of Mortality

Some colonies were apparently more successful than others, and there were distinct differences between the early and late sampling periods (Table 1). Survival was generally higher during the early (1981–87) sampling period. Some colonies (e.g., Bell Creek and Butler Peak) had relatively high survival (>90%) and low annual coefficients of variation (<30%). However, the reverse was not true. Colonies with relatively low survival (i.e., West Green and Green NW) generally had high coefficients of variation (62–74%). High CVs suggest that mortality was exceptionally high in some years (episodic) rather than chronic.

A similar pattern was evident during the late (1991–97) sampling period. A few colonies (e.g., Mount Franklin and Sherk Lake) had relatively consistent survival rates (CV <25%) with no suggestion of high-mortality events. However, most colonies suffered dramatic changes in survival from year to year (CV >50%). For example, the Bell Creek natural colony exhibited moderate survival (50–100%) in most years, but lost most of its marmots in a single year (1995). Data from the well-known Haley Lake colony are also illustrative. At this colony, 3 of the 4 survival years since 1994 represented high-mortality events (<50% survival). Given the natural history of *M. vancouverensis*, no colony could be expected to withstand such losses within a 4-year period, and this was the fundamental cause of near-extinction at this site by 1998.

Temporal Incidence of High-Mortality Events

High-mortality events were apparently not uniformly

distributed across time (Fig. 2). Annual survival data coded as high-mortality "events" or low-mortality "non-events" using a variety of criteria suggest that the frequency of events increased over time. Using the median survival rate (65%), results suggest a gradual increase in the event:non-event ratio. Other event criteria produce different results. Using the high survival event criterion (<80%), results suggest that marmots have generally not experienced high survival since the mid-1980s.

Of particular interest are the results based on a more stringent (<50%) event criterion. Episodes of high mortality occurred often (60 events and 141 non-events; rate = 30%). In 9 years the event:non-event ratio was higher than this, and in 10 years it was lower. The data suggest a "normal" event:non-event ratio of 20–25%. The ratio has been higher than 35% since 1994, and it was also higher than this during the 1984–86 period. The early sampling period is interesting because it does not suggest high spatial correlation. Inspection of the data suggests that events occurred at widely separated colonies (e.g., Mount Whymper, "P" Mountain, Mount Buttle, Green Mountain) during the mid-1980s. It is also interesting that these events went largely unnoticed by wildlife managers, perhaps because attention was focused on the expanding clearcut colonies at that time.

Spatial Correlation of Survival

Marmot survival was spatially autocorrelated during both sampling periods (Fig. 3). Moran's *I* varied from 0.09 to 0.52, indicating weak to moderate positive correlation at all effects radii. The strength of the correlation declined with increasing distance. Significant positive correlation was obtained at the largest radii that encompassed the entire study area (15 km), but strengths of the associations were low (*I* = 0.09 and 0.25). At shorter distances (<5 km) stronger correlations were obtained (*I* = 0.15 to 0.36).

Spatial autocorrelation differed between the early and late periods although the shapes of the curves were similar. Survival rates were more highly correlated during the late sampling period. This is an interesting result because recent extinctions ensured that the metapopulation was more dispersed in the late period than in the early period.

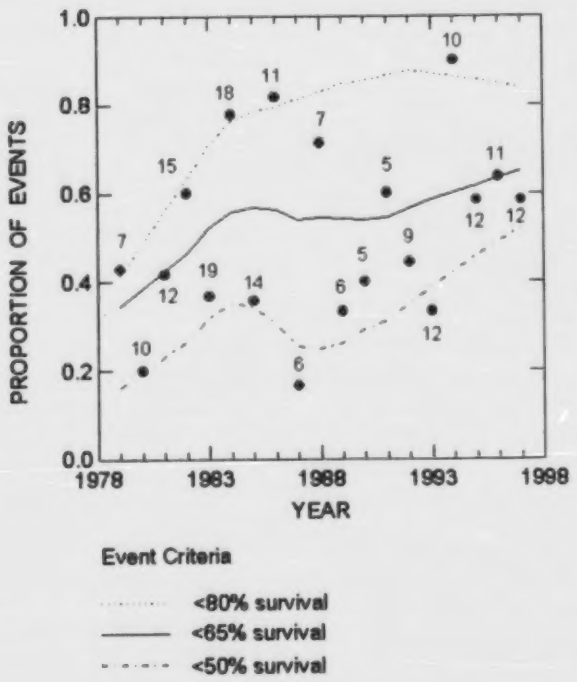

Figure 2. Increasing incidence of high-mortality events. Data points are event:non-event ratios using median annual survival (65%) as the event criterion. Lines are LOWESS regressions using 3 event criteria (50%, 65%, and 80%, as described in the text). Numbers of colonies in the sample are shown. Vancouver Island marmots have suffered from increased frequency of high-mortality events in recent years.

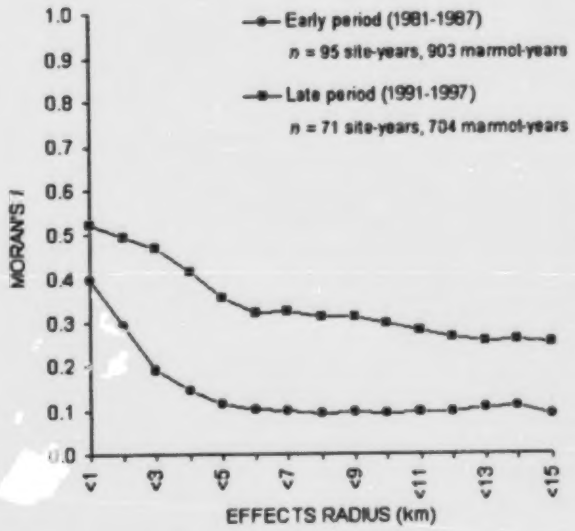

Figure 3. Spatial autocorrelation of marmot survival. Data are Moran's I coefficient using annual survival rates weighted by the numbers of adults and pups present. All correlations were significant and positive ($Z > 1.65$, $P < 0.05$). The degree of spatial autocorrelation increased between the early and late sampling periods, despite a more dispersed population structure.

DISCUSSION

There is little evidence in support of the hypothesis that marmots have recently responded to natural habitat change. Only 2 natural colonies (Westerholm basin and Green NW ridge) showed consistently poor survival, as would be predicted under a hypothesis of post-fire succession or tree invasion. In clearcut habitats, several sites (e.g., Pat Lake, G2C, Mount Franklin) showed relatively low and consistent survival. These results are congruent with a hypothesis of low-quality "sinks," particularly among habitats created by clearcut logging of high-elevation forests (Bryant 1996).

The apparent increase in frequency of high-mortality events is disturbing, and this trend appears to be at the root of recent population declines. Many apparently large and well-established colonies in natural subalpine meadows suffered abrupt and catastrophic mortality events over a short period ("P" Mountain, Mount Whymper, Gemini Peak, Bell Creek, Haley Lake). This pattern is inconsistent with a suggestion of gradual vegetation changes because habitats are not likely to change simultaneously. Moreover the temporal pattern of high-mortality events is inconsistent with the "bad weather" hypothesis because high-mortality events were not restricted to particular years.

Nearby colonies displayed similar survival rates. Significant positive spatial autocorrelation is consistent with a hypothesis of localized predator effort or disease events, but it is inconsistent with the suggestion of "bad weather" years or changing natural habitats. Most importantly, increased magnitude of spatial autocorrelation in the later sampling period suggests growing importance of mortality factors such as disease or predation. Neither predation nor disease can be divorced from forestry activities. It is likely that the importance of both factors has been increased by the changes in population density produced by marmot colonization of clearcuts.

Vancouver Island marmots illustrate many of the practical difficulties involved in applying Caughley and Gunn's (1996) approach to a real endangered species issue. Some difficulties were unavoidable. For example, funding limitations produced high variance in survey effort among years. Other difficulties were avoidable. Specifically, despite substantial count efforts in some years, results were often biased by uneven sampling. Interpretation remained difficult because results were not mapped, analyzed, or tested for consistency. The result was that as recently as the early 1990s there remained considerable uncertainty about whether marmots were increasing or declining (Janz et al. 1994).

Superficial analysis of population counts without benefit of demographic knowledge or spatial context led to the mistaken impression that marmots were thriving in clearcuts. Similarly, much speculation was made about issues such as dogs, ski-hill development, all-terrain vehicles, weather, natural habitat change, and changing predator abundance (e.g., Merilees 1980, Smith 1982, Dearden and Hall 1983, Milko 1984). There is nothing wrong with making such hypotheses, but unfortunately they were not accompanied by testable predictions. The result is that several ideas gained widespread acceptance without a shred of evidence to support them.

Population surveys can provide much valuable information with which to pursue what Caughley and Gunn (1996) called the "diagnostic" phase of endangered species management. However, simple measures of abundance will only be useful if the basic biology of the organism is understood, if proper sampling design is employed, and if results are tested for consistency. Conversely, poorly designed surveys in the absence of basic life-history knowledge may provide highly misleading results. Most importantly, hypotheses unaccompanied by testable predictions may provide excuses for inaction or encourage inappropriate management activities.

ACKNOWLEDGEMENTS

Many people participated in marmot counts, including K. Atkinson (deceased), D. Blumstein, C. Bryant, L. Campbell, J. Daniels, M. deLaronde, R. Davies (deceased), D. Doyle, L. Dyck, K. Fry, V. Hiensalu, D. Janz, F. Lockwood, J. Lewis, M. Loedel, B. Mason, G. MacDermott, K. McDonald, D. Milne, J. Morgan, D. Pemberton, J. Pendergaast, C. Ramsay, G. Smith, K. Sturmanis, W. Swain, J. Voller, L. Wilson, and M. Wong. K. Armitage provided helpful comments on the manuscript. I am greatly indebted to the Vancouver Island Marmot Recovery Team, who indulged my desire to spend a decade chasing obscure ground squirrels. The views in the paper are mine and may not necessarily reflect those of the Recovery Team.

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Ranking Potential Conservation Areas by Their 'Irreplaceability': Fish in Lakes in British Columbia, Canada

D.M. Forsyth

Centre for Biodiversity Research
Department of Zoology, University of British Columbia
6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada
Forsyth@zoology.ubc.ca

A.R.E. Sinclair

Centre for Biodiversity Research
Department of Zoology, University of British Columbia
6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

Determining sites of high conservation value is an important step towards developing an efficient network of protected natural areas. We applied a recently developed measure of conservation value, termed irreplaceability, to the problem of fish conservation in 2,159 lakes throughout British Columbia. The current (January 1998) protected area network included 292 of these lakes, but these protected lakes contained significantly fewer non-listed populations than would be expected if lakes were randomly selected for reservation; this likely reflects the ad hoc process of reserve selection that has occurred in British Columbia. However, since most lakes had low species richness (2), with few lakes containing many species, the average irreplaceability of lakes was very low (<0.4) under a range of conservation targets. Hence, the addition of a few key lakes to the present network of protected areas would achieve reasonable representation of most indigenous species. Only 17 of the 2,159 lakes were required to achieve representation of all fish species once but when the 292 lakes currently in protected areas were included as mandatory reservations, then 302 lakes were required to achieve the same goal. Devaluing stocked and non-native populations of fish in the calculations did not significantly alter the selection because most such populations occurred with few other species. Irreplaceability is thus a useful means of determining the value of potential reserves when conservation goals are complex.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Habitats for Tomorrow: Understanding the Consequences of Today's Decisions and Natural Disturbances on Future Habitat Condition

Walt Klenner
British Columbia Ministry of Forests
515 Columbia Street, Kamloops, BC, V2C 2T7, Canada
Walt.Klenner@gems7.gov.bc.ca

Russ Walton
#20-600 Cambridge Crescent
Kamloops, BC, V2B 5B6, Canada

Werner Kurz
ESSA Technologies Ltd., Suite 300, 1765 West 8th Avenue,
Vancouver, BC, V6J 5C6, Canada

ABSTRACT

We present the results of a study to examine the role of management actions and natural disturbances on the evolution of habitat patterns in forested ecosystems. Management strategies to maintain mature and old-growth forest in old-growth management areas need to consider the role that natural disturbances such as wildfire play in compromising plans to maintain such special habitat features. Old growth in the study area landscape unit continued to increase in lodgepole-pine-dominated forests for 50 years regardless of the management system implemented, because of the current large amount of mature forest. Douglas-fir forest types take longer to reach old-growth status, and will not reach target levels for at least 100 years, again regardless of the management system used. Even under modest assumptions of natural disturbances that used only wildfire at 25% of recent historic levels, up to 50% more area may need to be set aside as old-growth management areas to achieve and maintain long-term target levels of old-growth. Unless more diverse harvesting practices are adopted, high levels of fragmentation with few large patches of similar aged forest are likely to be present within 50 years. Aggregating harvest blocks into larger units is 1 approach that will help maintain a more diverse patch size distribution.

Key words: fragmentation, landscape ecology, landscape planning, old growth, simulation models.

Maintaining biological diversity has become an important land management issue internationally and in local jurisdictions (Wilson 1988, Fenger 1996). General guidelines have been established for habitat indicators such as the distribution of forest seral stages, the patch size distribution of cut and leave areas, riparian management practices (B.C. Forest Service and B.C. Environment 1995), and the retention of critical habitat structures such as residual live trees, snags, and downed wood (Franklin and Spies 1991, Franklin et al. 1997). These practices attempt to emulate a component of the habitat structures and patterns that characterize unmanaged landscapes (Hunter 1993).

Maintaining habitat complexity in forests managed for multiple resource objectives represents a strategic, "coarse filter" approach to habitat management (Hunter 1990) that complements other initiatives such as managing for individual species at risk where necessary. Managing for thousands of species on an individual basis is unwieldy and in many cases impossible since the life history requirements of most species are poorly known. It has been estimated that "coarse filter" habitat management may be sufficient to maintain 85–90% of species and the ecological processes that sustain them (The Nature Conservancy 1982, Noss 1987). Hence, it is critical that habitat management objectives be clearly defined and suitable approaches implemented to ensure targets are met.

Two concerns about conventional forest management practices—the loss of old-growth habitats and habitat fragmentation—have received much attention in the recent literature (Harris 1984, Mladenoff et. al. 1993, Spies et al. 1994). Although they can be treated separately, these 2 issues should also be considered together since large patches (e.g., >200 ha) of old forest are important to some species because of the forest interior conditions these habitats provide (e.g., special microclimate conditions), or the large spatial requirements of some species (Harris 1984, Shafer 1990,

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Angelstam 1992). In British Columbia, publicly owned forested lands are being divided into large (20,000–200,000 ha) "landscape units" for which strategic habitat management objectives have been established (B.C. Forest Service and B.C. Environment 1995). Targets for seral stage distribution, the patch size distribution of cut and leave areas, and the retention of old-growth forests across diverse ecosystems are being addressed. This shift towards targets for old growth and acceptable levels of fragmentation is proving to be a challenge to resource managers and planners. The long-term implications of management actions are not always clear, especially when a large proportion of the current landscape is mature or old-growth forest, and social or economic impacts must also be considered. This task is exacerbated by the interaction between managed disturbances and stochastic natural disturbances for which the probability of occurrence may be well known, but the location remains uncertain. In this paper, we explore the utility of 2 management approaches for addressing the seral stage and patch size characteristics of forested habitats: 1) the use of old-growth management area (OGMA) reserves to achieve desired old-growth targets; and 2) the use of aggregated harvesting approaches to reduce fragmentation and maintain a diverse range of patch sizes.

METHODS

General Description of the TELSA Model

The TELSA model (Tool for Exploratory Landscape Scenario Analyses; Klenner et al. 1997) was used to project current forest characteristics under a range of forest management and natural disturbance scenarios to evaluate the future habitat conditions that would likely develop. TELSA is a spatially explicit model that incorporates a detailed forest stand structure simulator (VDDT; Beukema and Kurz 1995), stochastic natural disturbances, user-specified management actions, and a diverse range of spatial and non-spatial indicators to evaluate landscape scenario projections. Multiple scenarios, and multiple iterations of the model within a scenario are used to assess the effects of stochastic natural disturbances and user-defined management actions in user-defined combinations. Only wildfire was considered in the present analyses, with the frequency and extent of wildfire disturbances, and consequences to stand structure and succession calibrated to local conditions.

TELSA Model Conditions

The TELSA model was applied to a 62,966-ha study area in British Columbia known as the Tranquille landscape unit (Fig. 1). Approximately 74% of the Tranquille landscape unit is forested (46,300 ha) and most of the remainder is classified as open grassland. Over half (26,133 ha) of the forested land is dominated by forests that have developed under a disturbance regime characterized by frequent stand-maintaining fires [(primarily low-intensity underburns in ponderosa pine (*Pinus ponderosa*) or Douglas-fir (*Pseudotsuga menziesii*) stands, (NDT4; B.C. Forest Service and B.C. Environment 1995)]. The remainder of the area is dominated by forests that historically have had frequent stand-replacing fires (NDT3), primarily crown fires in lodgepole pine (*P. contorta*), Engelmann spruce (*Picea engelmannii*), or subalpine fir (*Abies lasiocarpa*). Stand development in relation to the management practices and natural disturbances in the Tranquille landscape unit was calibrated by a team of plant ecologists and silviculture researchers familiar with the ecology of the area. For the purposes of the analyses presented here, wildfire was the only natural disturbance agent considered.

Maintaining Old Growth

We developed simulations to assess 3 approaches to managing for a 15% old-growth seral target within the forested landscape unit: 1) no OGMA reserves, with harvesting acceptable across the entire land base except in nonforested areas; 2) 15% of the land base designated as no-harvesting zones; 3) 21.5% of the land base designated as no-harvesting zones. There were 40 and 47 of these OGMA zones (options 2 and 3, respectively) derived from a draft management plan for the study area. The OGMAs ranged in size from 9.5 to 1,034 ha (average of 217 ha) and were centred around areas that currently have a high proportion of mature or old-growth

Figure 1. Location of the Tranquille landscape unit study area in south-central British Columbia.

Table 1. Summary of the scenario conditions used to simulate long-term changes in the amount of old-growth in the study area.

Scenario code	% area as OGMA[a]	Wildfire: % of historic level
SF	0	25
H	0	100
15M	15	0
15F	15	25
21M	21.5	0
21F	21.5	25

[a] old-growth management area

Table 2. Summary of the scenario conditions used to simulate long-term changes in the patch size distribution in the study area.

Scenario code	No. of years between aggregated harvesting	Harvesting	Wildfire: % of historic level
H	n/a	no	100
S	n/a	yes	0
A5	5	yes	0
AF	5	yes	25
A2	2	yes	0

forest. Each of these scenarios was projected under 2 wildfire scenarios: 1) no wildfires; 2) wildfires at 25% of the historic level (Table 1). In addition, an historic wildfire scenario without harvesting was implemented as a reference against which other scenarios could be compared. Forests became eligible for harvesting when they reached 100 years of age in each of the above scenarios. Four Monte Carlo simulations were conducted for all scenarios that involved wildfire, and mean values for the 4 simulations are presented. Management-only scenarios are deterministic and showed little spatial variability between runs.

Aggregated Harvesting and Patch Size

We developed simulations to harvest timber using either a conventional dispersed cutblock system or a dispersed cutblock system with periodic aggregated entries. We employed 15- to 40-ha clearcuts across the entire study area, and established a 300-m adjacency buffer around harvested openings with potential harvest blocks in this zone ineligible for harvest until the harvested stand reached 3 m in height (17 years). All forests became eligible for harvesting when they reached 100 years of age. Seventy-two potential aggregated harvesting areas ranging in size from 242 to 1,435 ha were established over the study area. When an aggregated area was selected for harvesting, all mature forest >100 years old was harvested with the intent of creating 1 large opening. Aggregated harvest area locations were selected to harvest the most fragmented of the 72 candidate areas, based on the highest proportion of the area in patches of <40 ha.

We developed 5 scenarios (Table 2) for comparison, including an historic wildfire scenario with no harvesting, which was calibrated to the estimated frequency and size distribution of historic wildfires in NDT3 and NDT4 forests. A target annual harvest level of 500 ha was established for these scenarios. When a large aggregated harvest area was chosen and the mature forest area exceeded the annual target harvest, the remainder was harvested the following year before continuing with dispersed harvesting.

We used 4 age categories to classify the seral condition of habitats: early, young, mature, and old growth. In NDT3 forest types, the respective age categories were 0–40, 41–100, 101–140, and >140 years. In NDT4 forest types, the respective age categories were 0–40, 41–100, 101–250, and >250 years. A patch was defined as an area within the above age categories, and within the same biogeoclimatic zone. This approach does not reflect the complex biological issues that may need to be considered in defining a patch, but represents an index used to evaluate current conditions and to assess changes over time.

RESULTS

Maintaining Old Growth

The ability to maintain a 15% target level of old-growth forest in the study area was a function of initial habitat conditions, assumptions about future wildfire conditions, and the amount of area reserved from harvest. NDT3 habitats in the study area are currently in a non-equilibrium condition stemming from a period of high wildfire activity about 100 years ago. A large amount of mature habitat will be recruited into old growth in the near future (Fig. 2a), and the 15% target is easily exceeded regardless of management treatment. Beyond 100 years, management strategies play an important role in determining the amount of old-growth forest. In contrast, NDT4 habitats (Fig. 3a) currently exhibit a shortage of old growth and this continues for approximately 100 years, again regardless of management treatment. Under an historic wildfire regime and no harvesting, approximately 30% of the forest within the landscape unit became old growth in NDT3 habitats by year 125, while in NDT4, the levels continue to increase until a peak is reached in year 150 (27%). The current condition of the landscape, in combination with the difference in age when old growth is achieved (141 years for NDT3, 251 years for NDT4) plays an important role in determining the long-term dynamics of old-growth forest over time. In NDT3, 14.9, 20.2, 48.9, and 16.0% of the study area is currently in the early, young, mature, and old-growth categories, respectively, whereas in NDT4, the proportions are

6.9, 11.9, 78.0, and 3.2% respectively.

In both NDT3 and NDT4 habitats, less than half of the 15% old-growth target developed outside of the OGMA reserves, illustrating the need for special management areas to maintain old forest habitat. Following the recruitment phase (Figs. 2b and 3b), only about 7% and 4% of the area outside of the OGMA develops into old growth in NDT3 and NDT4 forest respectively. The proportion of old growth in the study area contributed by OGMA reserves increases as expected for both NDT3 and NDT4 habitats for the 2 scenarios without natural disturbances. However, when wildfire is incorporated into the management scenario at 25% of recent historic levels, 25–30% of the OGMA is maintained in an earlier seral condition. A 50% increase in the area designated as OGMA (from 15% to 21.5%) is necessary if the 15% old-growth target in the study area is to be maintained. In landscapes where most of the land base is available for harvest, or where inoperable land does not support suitable old-growth forest, OGMAs will likely be 1 of the only sources of old-growth forest. If wildfires in the study were to increase to historic levels (for example, as a consequence of short- or long-term climate change), even a 50% increase in the area being maintained as OGMAs (21.5%) would not achieve the desired 15% old-growth target.

The loss of old growth from OGMAs due to wildfire, and the limited potential for recruiting old growth from outside the OGMAs (Figs. 2b and 3b) indicates that maintaining old growth solely on the limited OGMA land base is a risk-prone strategy. Furthermore, since harvesting targets stands as they reach the early mature condition (100 years), recruiting old-growth forest from the harvested land base to replace old growth lost in an OGMA may be delayed by anywhere from 40 to 150 years (NDT3 habitat, mature at 100, old growth at 141; NDT4 habitat, mature at 100, old growth at 251).

Aggregated Harvesting and Patch Size

We examined changes in the patch size distribution of early and young forests in relation to the patterns created by historic wildfires and 4 other management scenarios (Table 2). Although much (>50%) of the mature and old-growth forest in the current landscape is in large patches (251–1,000 ha), there are few large patches of early and young seral forest

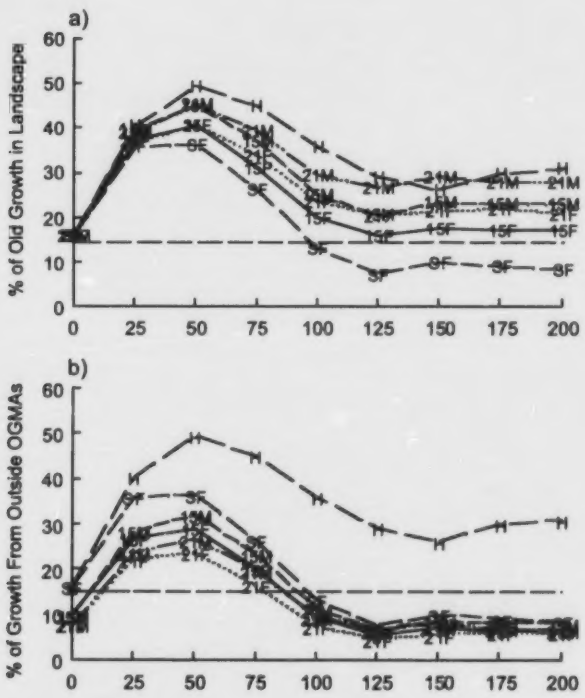

Figure 2. Changes in the amount of old-growth forest in NDT3 habitat in the Tranquille landscape unit, (a) across the entire landscape unit, and (b) outside old-growth management areas (OGMAs). Symbols as indicated in Table 1.

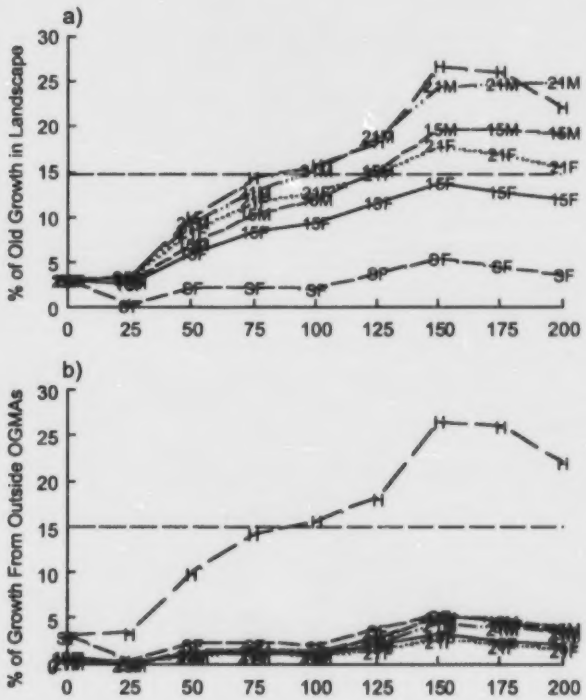

Figure 3. Changes in the amount of old-growth forest in NDT4 habitat in the Tranquille landscape unit, (a) across the entire landscape unit, and (b) outside old-growth management areas (OGMAs). Symbols as indicated in Table 1.

(Fig. 4). Standard dispersed harvesting with small clearcut openings provides no opportunity to create large patches of similarly aged forest. Aggregated harvest entries at 5-year intervals maintain a component of large patches in the landscape approaching the levels created by historic wildfires, while increasing the frequency of aggregated harvesting to every second year (Fig. 5) provides a greater amount than created by the historic wildfire conditions modelled here.

DISCUSSION

The scenario analyses we conducted illustrate the need for developing special resource management zones to maintain critical habitats such as old-growth forest, and special harvesting approaches to reduce habitat fragmentation. The inclusion of wildfire in our simulations illustrates that natural disturbances may interfere with management objectives, indicating the

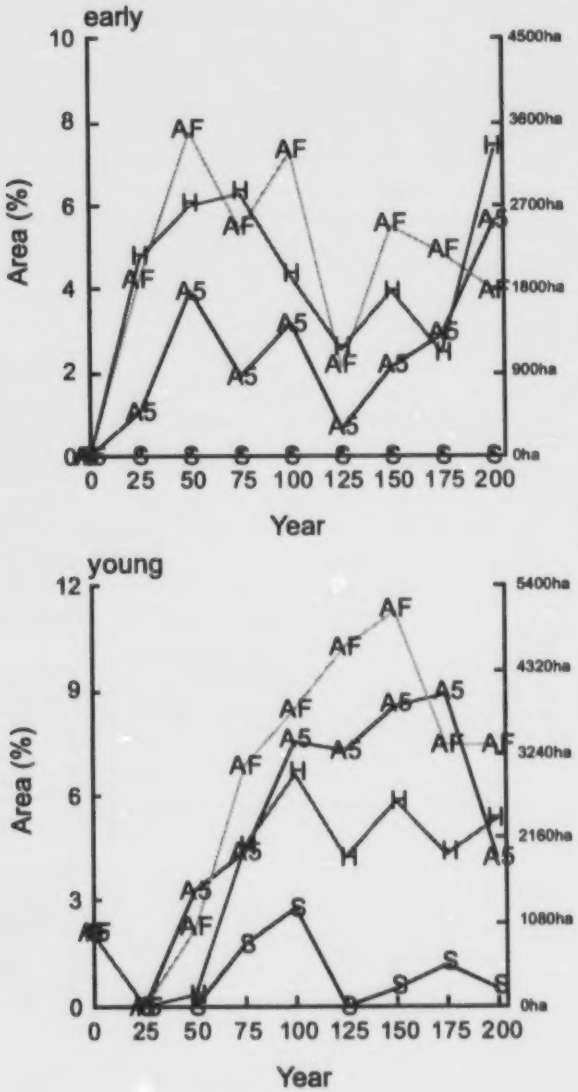

Figure 4. Changes in the amount of habitat in large patches (251–1,000 ha) of (a) early seral habitat, and (b) young forest in relation to harvesting pattern strategy. Symbols as indicated in Table 2.

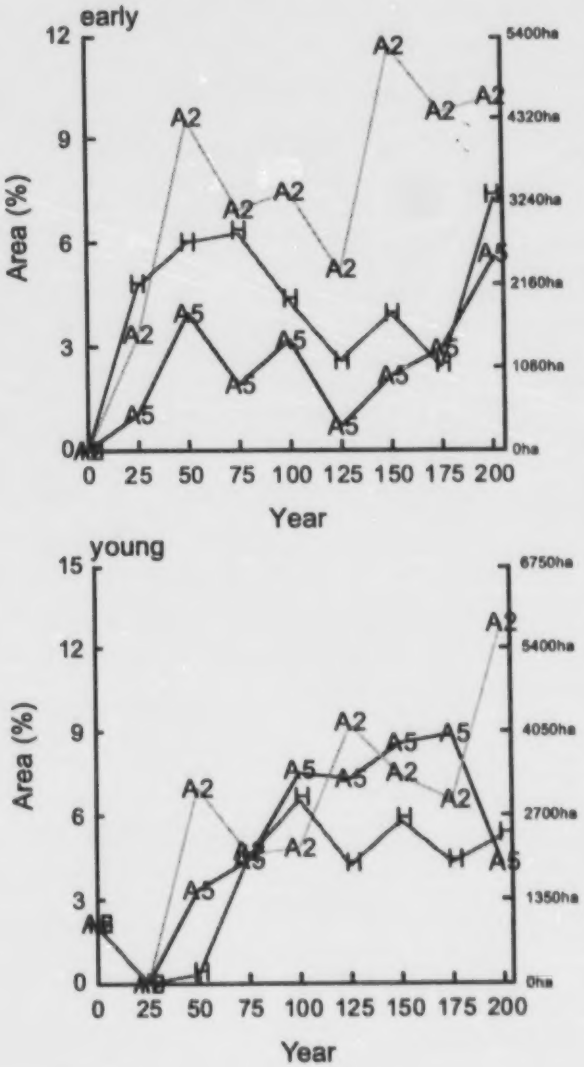

Figure 5. Changes in the amount of habitat in large patches (251–1,000 ha) of (a) early seral habitat, and (b) young forest in relation to the frequency of aggregated harvesting. Symbols as indicated in Table 2.

need to implement robust management approaches that diminish the likelihood that desired objectives are not entirely compromised by a chance event. Yet unexpected wildfires do occur and will likely continue to occur. In our simulations, historic levels of wildfire affected, on average, 525 ha per year, but annual variability was high. Assuming that fire suppression in managed forests will be somewhat effective in diminishing both the number and size of wildfires (e.g., to 25% of historic levels), high fire years in which >1,000 hectares burned within the study area still occurred.

Managing for desired landscape objectives will require some redundancy in management actions in the event a catastrophic natural disturbance reduces or eliminates opportunities for maintaining specific values. Landscape analysis and planning approaches (e.g., Diaz and Apostol 1992) need to consider the inevitable natural disturbances that will occur over long time periods, and develop contingencies. Too great a reliance on maintaining a specific feature may at best be highly uncertain, or at worst, detrimental to planning for long-term objectives, as it creates a false sense of security about future landscape condition. Landscape habitat modelling tools (e.g., Hansen et al. 1993, Klenner et al. 1997, McCarter et al. 1998), when used in concert with landscape analysis and design, can play an important role in exploring options for dealing with uncertainty. Multiple scenario projections to assess the efficacy of different plans for achieving targets can then be translated into planning scenarios where a full complement of forest values is evaluated. In the case of the Tranquille landscape unit, the loss of old-growth forests in OGMAs due to wildfires is compounded by the lack of old-growth recruitment potential from the non-OGMA land base. However, redundancies in landscape planning, such as setting aside greater amounts of old-growth management area reserves than required at the time of implementation (e.g., establishing 21.5% of the landscape vs. the 15% target) to anticipate the future loss of old-growth forest to natural disturbances also reduced the level of timber harvested. Where such impacts on timber supply are unacceptable, extended rotation (Curtis 1997) management zones may provide opportunities for maintaining some old-growth, late mature timber for recruitment into old-growth, and timber harvesting.

Forest harvesting with small cutblocks has been criticized because of the high levels of fragmentation associated with this approach (Harris 1984, Franklin and Forman 1987). Li et al. (1993), Wallin et al. (1994), and Gustafson and Crow (1996) and have clearly demonstrated that larger cutblocks, or harvesting in constrained zones without adjacent leave strips can help reduce the level of fragmentation caused by harvesting. We have extended this concept to an application in the Tranquille landscape unit, and have made a preliminary assessment of the frequency that aggregated harvest entries need to be initiated to approximate the patterns expected from historic levels of wildfire. The study area we chose for our simulations has been harvested for approximately 30 years, primarily around several "nodes" of more concentrated activity. This pattern facilitated maintaining a diverse range of patch sizes using periodic aggregated harvest areas since a high proportion of the mature and old-growth forest in the landscape consisted of very large patches (i.e., 50% >251 ha). Had the study area contained more widely dispersed cutblocks, there would be fewer opportunities for developing large patches of similarly aged forest (Wallin et al. 1994).

We found that aggregated harvesting implemented every 2–5 years should maintain a diverse range of patch sizes in the landscape. The frequency of aggregated harvesting entries will need to be reconciled with broader resource objectives (e.g., water resources, visual impacts, recreation, etc.), implemented on a landscape or habitat-specific basis, and revisited periodically to evaluate the landscape condition in light of the cumulative effects of harvesting and natural disturbances. We did not explicitly track information on the amount or dispersion of roads in the present study, but roads and access management are important habitat management issues (McLellan and Shackleton 1992) that need to be considered. Dispersed cutblocks require an extensive and permanent network of roads to access timber on an ongoing basis. Roads can have negative impacts on some species by presenting barriers to dispersing organisms, forming corridors along which invading organisms (e.g., weeds) enter an ecosystem, increasing the incidence of human-caused fires, and facilitating excessive legal hunting pressure and/or poaching by providing access (Thomas et al. 1976, McLellan and Shackleton 1992, Thurber et al. 1994). From the perspective of maintaining large patches of habitat and diminishing roads and access, aggregated harvesting entries should be planned around areas that currently have a high level of roads before new areas are developed.

ACKNOWLEDGEMENTS

We acknowledge the support of Forest Renewal British Columbia and the British Columbia Ministry of Forests in funding the development and application of the TELSA project. A. Arsenault, I. Cameron, D. Lloyd, L. MacLaughlan, H. Merler, and A. Vyse helped develop and review successional characteristics and natural disturbance parameters for forest types in the Tranquille landscape unit. Planning and technical services staff at the B.C. Ministry of Forests, Kamloops District helped prepare the database for the Tranquille landscape unit.

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Using Selection to Define Crucial Caribou Habitat: A Case For Caution

Robert B. Anderson[1]
Department of Biological Sciences, University of Alberta
Edmonton, AB, T6G 2E9, Canada

Elston Dzus
Eco Edge Inc, c/o Boreal Caribou Research Program,
15810-114 Avenue, Edmonton, AB, T5M 2Z4, Canada

Stan Boutin
Department of Biological Sciences, University of Alberta
Edmonton, AB, T6G 2E9, Canada

EXPANDED ABSTRACT

Key words: Alberta, habitat, *Rangifer tarandus caribou*, woodland caribou.

Woodland caribou (*Rangifer tarandus caribou*) are on Alberta's Blue List of species that may be at risk of declining to non-viable population levels in the province, and are designated as Threatened under the provincial wildlife act (Alberta Wildlife Management Division 1996, Alberta Environmental Protection 1998). Concern for this species led to the development of Alberta's *Procedural Guide for Petroleum and Natural Gas Activity on Caribou Range*. These guidelines allow for industrial activity in caribou range, "provided the integrity of the habitat is maintained to support its use by caribou" (Alberta Energy 1991). To manage for habitat integrity we must be able to identify features of high quality caribou habitat. Bradshaw et al. (1995) began to address this need by examining winter habitat selection at a coarse scale (1:250,000). They found that habitat polygons dominated by treed fens and bogs were selected by boreal woodland caribou; caribou avoided areas with little or no peatland habitat. Bradshaw et al. (1995) suggest that large areas of treed-peatland-dominated habitat are crucial to the survival of caribou in northeastern Alberta. Upland-dominated areas are considered unsuitable habitat (Stuart-Smith et al. 1997). We tested these conclusions by examining winter habitat use and selection for an adjacent, previously unstudied, caribou population. At the coarse scale, compositional analysis (Aebischer et al. 1993) was used to determine whether caribou in the Red Earth range select areas dominated by treed fens and bogs over areas dominated by upland habitat. Unlike animals in Bradshaw et al.'s (1995) study area (Athabasca), the Red Earth population did not select areas dominated by treed fens and bogs over upland-dominated areas. However, variation was high among Red Earth caribou. This variation was explained when the Red Earth population was split into a southern subpopulation and a northern subpopulation. The northern subpopulation selected areas dominated by treed fens and bogs over areas dominated by upland habitat, as was expected from the conclusions of Bradshaw et al. (1995). However, the southern subpopulation acted in a converse manner, selecting areas dominated by upland habitat over areas dominated by treed fens and bogs. This unexpected result was further investigated by examining habitat use at a finer scale (1:20,000). Fine-scale analysis revealed that all caribou in both the Red Earth and Athabasca regions of Alberta used treed peatland stands most commonly, regardless of the dominant habitat at the coarse scale.

These results indicate the need for caution when habitat selection is analyzed and conclusions are made about the crucial nature of a given habitat class. It is important for researchers to realize the limitations of the mapping scale they are using for use and selection analyses. This study demonstrates that mapping scale can have a large impact on results. If this study had been done only at the coarse scale, we may have concluded that peatlands were not as important as was first believed. However, by going to a finer scale we were able to see that even caribou in upland dominated areas use peatland stands extensively. The results of this study also suggest that selection alone is not a good indicator of the critical nature of a given habitat class. Although Bradshaw et al.'s (1995) results were very strong, our discovery of woodland caribou in an area with very few polygons dominated by treed fens and bogs raises questions about the "crucial" nature of that landscape composition. It is important to remember, however, that occupation is not a measure of habitat quality, because occupied habitat may also be a population sink (Danielson 1992). Further study will be required to determine whether caribou in the southern portion of the

[1] Present address: Applied Ecosystem Management Ltd., 11301-89 Avenue, Grande Prairie, AB T8V 5X3, Canada. rba@telusplanet.net

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 498pp.

Red Earth range have similar population dynamics to caribou in areas where habitat dominated by treed fens and bogs is prevalent. Similar survival and birth rates in the different landscapes would suggest that large areas dominated by treed fens and bogs are not as crucial as first believed.

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Quantifying Aerial Photo Polygon Complexity on a Subalpine Landscape of Wells Gray Park, British Columbia

Wei Zhang
Botany Department, University of British Columbia
3529 - 6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada
weiwzh@interchnage.ubc.ca

Gary E. Bradfield
Botany Department, University of British Columbia
3529 - 6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

This study examines patterns of landscape diversity in a 20,000-ha montane-subalpine forest in southern Wells Gray Provincial Park, B.C. Landscape pattern was assessed by sampling aerial photos to quantify the extent of variation in topographic features and polygon (ecosystem unit) distribution patterns. Sixty-eight circular plots, representing pixel sizes of 450 m and 750 m diameter, were used to assess landscape properties, including terrain features and the types and arrangements of polygons. Vegetation data for the polygons were extrapolated from 170 ground plots sampled throughout the study area. Relationships between various complexity indices derived from the polygon patterns and physical terrain features were examined using Principal Components Analysis and Pearson correlations. Vegetation variation within polygons (beta-diversity), as defined by the first axis of Detrended Correspondence Analysis, was also examined for 5 main polygon types in relation to major abiotic factors and forest structural attributes. The results suggest that polygon complexity is weakly related to the interaction of elevation, slope, and aspect. The influence of natural disturbance from windthrow, fire, rock slides, and avalanches on polygon complexity may depend on the extent of disturbances and the conditions prior to the disturbance. Total length of polygon boundaries was the most influential variable on the polygon complexity and diversity indices tested. Overall beta-diversity within a polygon type was related to stages of vegetation development, hence, disturbance history. Beta-diversity of vegetation in the same stage of development was strongly influenced by physical environment, (e.g., elevation, slope, and aspect).

Key words: aerial photography, landscape diversity, subalpine.

Subalpine landscapes exhibit wide variety in physical terrain features and vegetation patterns. Recent advances in landscape ecology offer new approaches for quantifying pattern complexity from maps and remote imagery (Forman and Godron 1986, Krummel et al. 1987, Forman and Godron 1988, Turner and Ruscher 1988, Ritters et al. 1995, Metzger and Muller 1996, Rescia et al. 1997). In particular, a number of landscape complexity indices have been developed for describing relationships among spatial structures (e.g., forest/nonforest patches) and making inferences about how those relationships change through time (Krummel et al. 1987, 1988; Turner 1990; Sachs et al. 1998). Owing to the inherent variability of landscape systems, some indices have proved unsuccessful in demonstrating useful relationships (Hulshoff 1995, Garrabou et al. 1998). Other comparative studies have searched for factors in common among different indices (Ritters et al. 1995, Cain et al. 1997). Despite the availability of quantitative methods for description and forecasting at the landscape scale, knowledge of this type for most areas in British Columbia is lacking.

A subalpine landscape may be thought of as a complex mosaic of vegetation structural types dispersed over a highly variable physical terrain. The resulting patterns are commonly shown as polygons (i.e., ecosystem units) on forest cover maps. At the landscape scale, ecological questions may relate to the types, sizes, and distributions of polygons in relation to landform and disturbance. Alternatively, questions may be asked relating to the structure and dynamics of vegetation within polygons. The 2 approaches are complimentary, as each may identify processes affecting the other. For example, a set of adjacent polygons may be maintained by disturbances such as fire, rock slides, and avalanches. While many previous studies on subalpine vegetation have been reported, the use of polygons to organize and extrapolate information to the landscape scale has not been attempted in this ecosystem.

The objective of this study is to quantitatively describe the complexity of a subalpine landscape by: 1) examining relationships among polygon patterns (e.g., types, sizes, boundaries) and various environmental factors, including topography, soils, and disturbance; and 2) examining the

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1. Polygon attributes measured or calculated for the aerial photo plots. Range of values shown pertain to the larger (750 m diameter) plot size used.

General description and calculation	Range in large plots
Number of polygon types (NPT)	4–18
Total number of polygons (NOP)	4–27
Number of boundary types	3–28
Total number of boundaries (TNB)	3–40
Total length of boundaries (mm on aerial photo)	116–518
Percentage of plot area occupied by largest polygon	11.7–70
Shannon diversity index	0.86–2.61
Shannon evenness index	0.53–0.97
Simpson diversity index	1.92–12.56
Simpson evenness index	0.28–0.92
Polygon dominance index {DI = Ln[NPT] + $\Sigma[P_k Ln(P_k)]$, where P_k denotes proportion of polygon type k}	0.05–0.79
Shape index {LSI = $(0.25\Sigma L_{ik})/A^{1/2}$, where L_{ik} denotes total length of edge between polygon type i and k}	0.65–2.92
Average perimeter:area ratio	0.06–0.26
Area complexity index {ACS = H(a) x H(c), where H(a) denotes Shannon diversity of area types, and H(c) = H(a)/Log_2(NOP)}	2.20–28.08
Boundary complexity index {BCS = H(b) x H(c), where H(b) denotes Shannon diversity of boundary types, and H(c) = H(b)/Log_2(TNB)}	1.43–55.15

extent to which vegetation variation within polygons can be accounted for by environmental conditions.

STUDY AREA

A 20,000-ha area in southern Wells Gray Provincial Park, B.C., was selected for study. The area consists of a broad, mountainous plateau with steep-sided valleys and moist, forested highlands at elevations of 1,300–2,100 m. The majority of the area has been classified as Engelmann Spruce–Subalpine Fir (ESSF) zone, with wc2 (wet cold) and vv (very wet and very cold) being the dominant subzones (Lloyd et al. 1990). Throughout the study area, the forests have experienced a variety of natural disturbances from lighting strikes, snow avalanches, windthrow, and rock slides (scree).

DATA COLLECTION AND ANALYSIS

Polygons and Physical Terrain Features

Following the guidelines in *Standards for Terrestrial Ecosystems Mapping* (RIC 1995), polygons were mapped onto 1:15,000, colour aerial photos of the study area. Polygons represent subjectively delineated areas that are relatively uniform in vegetation, topographic features, and soil conditions.

To characterize polygon patterns across the landscape, 2 sizes of concentric circular plots (3-cm and 5-cm diameter plots on the aerial photos, corresponding to approximately 450 and 750 m on the ground) were systematically sampled on the aerial photos. In total, 68 plots of each size, spaced at roughly 8-cm intervals between centres on the photos, were examined. The use of 2 plot sizes allowed for testing of the effects of "size of observation window" on the assessment of landscape complexity. Within the aerial photo plots a number of polygon attributes, such as total number of polygon types, and boundary types and lengths (Rescia et al. 1997), were visually determined and used to calculate various diversity/complexity indices (Table 1). Owing to the occurrence of logging in parts of the study area, 8 plots were removed, leaving a total of 60 plots for analysis.

Also recorded from the aerial photo plots were a number of physical terrain features and disturbance conditions (Table 2). The physical data were obtained from inspection of 1:20,000 and 1:10,000 topographic (TRIM) maps of the study area. The types and extents of disturbances for each plot were determined through a combination of aerial photo interpretation and referencing an extensive database of the study area (Bradfield et al. in prog.). An aspect index (AI) for each plot was calculated as an area-weighted value of aspect favourability (Beers et. al. 1966). AI varies from –1.0 to +1.0, with higher index values indicating higher potential incident radiation.

Correlations within and between the sets of polygon variables and the physical terrain/disturbance variables were examined in 2 ways: 1) by calculating Pearson correlation coefficients (or Spearman correlations, depending on the nature of the variables); and 2) by applying principal components analysis (PCA) to each set of variables, followed by calculating between-set correlations using the first PCA axis as a summary descriptor of each set.

Table 2. Physical terrain features and disturbance variables recorded in the aerial photo plots.

Highest elevation (m)
Lowest elevation (m)
Elevation at the central point (m)
Within-plot elevation range (m)
Aerial coverage (%) of 5 slope categories: slope >100%, 50–100%, 20–50%, 10–20% and <10%
Slope at the central point (%)
Number of slope categories recorded in the plot
Aerial coverage of largest slope category (%)
Aerial coverage (%) of 9 aspect categories: N, NE, E, SE, S, SW, W, NW and Undulating/flat.
Number of aspect types recorded in the plot
Aerial coverage of largest aspect category in the plot
Aspect category at the central point (1–9)
Estimated area (%) weakly influenced by fire
Estimated area (%) moderately influenced by fire
Estimated area (%) heavily influenced by fire
Presence of windthrow (binary)
Aerial coverage (%) of rock outcrop
Aerial coverage (%) of avalanche
Aerial coverage (%) of scree/rockslide
Aerial coverage (%) of standing water
Aspect index (area weighted by aspect types)

Within-polygon Variation in Relation to Environmental Factors

Variation within polygons was assessed using information from a database of 170 vegetation plots sampled in an extensive ground survey of the study area during 1995–97 (Bradfield et al. in prog.). For this purpose, 5 polygon types that occurred frequently in the study area and contained a sufficient number of ground vegetation plots for analysis were selected (Table 3). Using this approach, minimum estimates of plant species richness were determined for each polygon type and for the smaller aerial photo plots (450 m diam) based on the polygons they contained. Difficulties in matching some polygons with ground vegetation plots impeded using this approach with the larger plots (750 m diam) on the aerial photos. Consequently, it was possible to examine the relationship between total plant species richness and landscape complexity.

Vegetation variation within the 5 main polygon types, separately and combined, was measured as the number of standard deviation (s.d.) units (termed beta-diversity) on the first axis of a detrended correspondence analysis (DCA) (Økland 1986, Backéus 1993, Zhang 1998). Two questions were addressed in analyzing the vegetation-environment relationships within the polygon types: 1) To what extent do individual environmental variables account for the observed species variation that is associated with beta diversity?; and 2) How successful are groups of related environmental variables in explaining the species variation? To answer the above questions, environmental factors were selected in decreasing order of their ability to explain the variance in species composition according to a Monte Carlo permutation test performed by CANOCO (version 3.15; ter Braak 1987, 1990). Twelve abiotic environmental variables in 3 categories (disturbances, topography, soil/surface features) were available for correlating with the species variation. In addition to examining relationships with abiotic factors, we also tested the abilities of various forest structural attributes, such as stand density, basal area, and canopy height, as well as the polygon type classification, to explain variation in species composition.

RESULTS AND DISCUSSION

Polygon Complexity, Physical Features, and Natural Disturbances

The aerial photo plots spanned an elevation range of 1,310–2,426 m and encompassed considerable variation in polygon features (Table 1). The range of calculated values for the Simpson index (SI) and the area and boundary complexity indices (ACS and BCS) was generally large, indicating that those indices could be sensitive to the differences among the plots of this study. Several correlations were detected among individual factors within each separate group (i.e., among the physical features and among the polygon features); however, there were no significant relationships between the variables of the 2 groups.

The first PCA axis of the physical terrain data explained 23 and 19% of the total variance for the large and small plots, respectively. For both plot sizes, the first PCA axis represented a terrain gradient from high elevations with steep slopes and N–NE aspects, to lower elevations with gentle slopes and S–SW aspects. The correlations between the PCA axis of the terrain data and the polygon attributes were weak; however, a weak but significant correlation ($r < 0.4$, $P < 0.05$) was shown for total length of polygon boundaries and the indices calculated from the length of boundaries.

The first PCA axis of the polygon data explained 61 and 63% of the total variance for the large and small plots, respectively. The axis represented an overall summary of the relationships among the area complexity index and several dominance indices, such as H and SI. There was no significant correlation between the first PCA axis from the physical terrain data and that from the polygon data.

The only significant relationship detected between polygons and terrain features was that between total polygon boundary length and the first PCA axis of the terrain data. The length of polygon boundaries is a function of the number and shapes of polygon patches on the landscape. Such boundaries are key features at the landscape scale, and have been linked to ecological processes influencing biotic and abiotic conditions that affect overall biodiversity (Forman and Godron 1988, Wiens et al. 1985, Holland et al.1991,

Table 3. Characteristics of the 5 main polygon types in the study area. Polygon letter codes refer to dominant understory species (FA, *Menziesia*; FV, *Valeriana*; FR, *Rhododendron*). Polygon numerical codes refer to developmental stages (4, saplings; 5, young forest; 6, mature forest; 7, old forest). "Variance explained" refers to variation in species composition accounted for by groups of abiotic (A–C) and biotic (D, E) factors.

	Polygon type (structural stage)					
	All	FA (4,5,6,7)	FV (5,7)	FR (4,5)	FR (6)	FR (7)
ESSF site[a]	wc2/vv	wc2	wc2/vv	vv	vv	vv
Number of vegetation plots	92	14	17	15	23	23
Species richness	156	68	79	63	87	91
Beta-diversity (s.d. units)	4.81	4.34	3.88	4.03	3.75	3.10
			Variance explained (%)			
Disturbances (A)	10	50	21	31	28	26
Topography (B)	10	28	23	32	34	31
Soil and surface features (C)	10	42	31	30	34	32
Forward selection from A+B+C	19	46	40	55	44	19
Vegetation structure (D)	20	86	67	77	58	60
Polygon types (E)	18	28	19	10		

[a] Engelmann Spruce–Subalpine Fir (ESSF) biogeoclimatic subzones: wc2 = wet cold; vv = very wet, very cold.

Metzger and Muller 1996, Rescia et al. 1997). Furthermore, our results suggest that polygon boundary features are also associated with a complex gradient of physical terrain factors and natural disturbances.

We suggest that the effects of disturbance on patterns of polygon complexity depend on the pre-existing patterns (i.e., before the disturbance occurred) and on the extent (i.e., total area affected) of the disturbance. For example, in a heterogeneous system, a disturbance such as a large fire may decrease landscape complexity. In contrast, in a homogeneous system, medium- to small-scale disturbances may lead to an increase in complexity. No effects related to the 2 plot sizes used on the aerial photos were detected in this study.

Within-polygon Vegetation Variation in Relation to Physical Environment and Disturbance

Of the 60 aerial photo plots examined, only 24 contained polygons that could be directly related to vegetation plots. Using that information, the minimum estimates of total species richness ranged from 21 to 70 species in the smaller (450 m diam) aerial photo plots. There was no significant relationship between the estimated species richness and polygon complexity or physical terrain features, although species richness was found to vary greatly within polygon types.

The vegetation beta-diversity ranged from 3.1 to 4.3 (s.d. units) for the 5 major polygon types examined (Table 3). The old-growth forest polygon type (FR7) showed the lowest beta-diversity, with polygon types containing mixed development stages (FA, FR, FV) exhibiting higher values of beta-diversity. The variance in vegetation data explained by different groups of environmental variables is also given in Table 3. The variation within polygon types with a single development stage (FR6 and FR7) tended to be related to topographic, soil, and surface features. Disturbance showed stronger impacts on species variation in the mixed polygon types, suggesting a closer relationship between stage of vegetation development, hence, disturbance history and species composition. The total variance explained by forward selection of significant variables ranged from 19% (combined polygons) to 55% (FR45). The low variance explained for the FR7 type indicates that other, unmeasured variables may be influencing old-growth vegetation. The effects of complex biotic interactions among species (e.g., host-pathogen, plant-mycorrhizal) may increase with developmental stage of the forest so that landscape variables become less influential.

The variance explained by forest structural attributes was relatively low (<30% from forward selection; Table 3), suggesting that overall forest structure and species composition are not strongly linked in the study area. Compositional variation of vegetation in the same stage of development was most strongly influenced by physical factors such as elevation, slope, aspect, soil, and surface features. In general, vegetation in multiple stages of development will be defined by mixed polygon types in the process of aerial photo interpretation. Therefore, within-polygon diversity will be largely influenced by variability in topographic features, soil conditions, and small-scale disturbances.

SUMMARY

- Polygon features recognized on aerial photos are weakly correlated with physical terrain features.

- The total length of polygon boundaries is the main determinant of polygon complexity and is the key factor related to physical terrain features.
- The impact of natural disturbances on polygon complexity depends on the extent of disturbance and the landscape conditions prior to the disturbances.
- Beta-diversity is related to the stage of vegetation development and environmental conditions.

ACKNOWLEDGEMENTS

The study was supported by Forest Renewal British Columbia (FRBC award #TO96045). We are grateful to B.C. Parks for permission to conduct research in Wells Gray Park. The aerial photograph interpretation and polygon mapping were done by K. Barton and G. Senichenko. Many individuals assisted with vegetation plot sampling including R. Botting, Dr. K. Marr, B. Marshall, J. Nagel, C. O'Brien, S. Peel, G. Senichenko, J. Schulte, P. Williston, and F. Wilson. We thank Dr. A. Arsenault and T. Goward for their inspiration throughout this study. The cooperation of the staff at the Wells Gray Park Education and Research Centre is gratefully acknowledged.

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Environmental Indicators for Natural Diversity

R.B. Smith
British Columbia Ministry of Environment, Lands and Parks
PO Box 9335 STN PROV GOVT, Victoria, BC, V8W 9M1, Canada
Risa.Smith@gems9.gov.bc.ca

ABSTRACT

As part of its *Environmental Indicator Series* British Columbia tracks information on five indicators of natural diversity: Species at Risk, Forest Species, Wildlife Populations, Fish and Toxic Contaminants in Biota. The indicators will be published, along with indicators for other environmental issues, in a document entitled *Environmental Trends for British Columbia.* Key indicators for each issue area will be presented, along with secondary measures that provide more detailed information. In summary, these natural diversity indicators show that: (a) freshwater fish are of particular concern in the province — over 35% of the forest dependent freshwater fish are threatened or endangered; (b) some wildlife which are managed for recreational purposes show significant declines in their ranges (i.e., mountain caribou, sharp-tailed grouse), while others show fairly stable ranges (i.e. black-tailed deer, moose); (c) changes in the distribution of 40% of forest-dwelling mammals is unknown; and (d) significant improvements in the concentration of toxic contaminants (DDE, PCBs and dioxin) in great blue heron eggs have occurred since the 1970s. In compiling the data for the indicator series some issues have arisen about the reliability of monitoring efforts. Ideally information for these indicators would come from a well-established monitoring network with full provincial coverage. Because the ideal situation seldom exists, information is compiled from several sources, including tracking by the Conservation Data Centre, inventories, research studies and expert opinion. The use of expert opinion can be a controversial source of data. However, it is sometimes the only data available for important indicators. It is expected that subjective information will be replaced, in the future, with more reliable monitoring data.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Flammulated Owl Habitat Model Validation for the Merritt and Kamloops Forest Districts

Astrid M. van Woudenberg
Cascadia Natural Resource Consultants Inc.
2675 Skeena Drive, Kamloops, BC, V2E 2M9, Canada
astrid@cnrc.com

David A. Christie
Cascadia Natural Resource Consultants Inc.
2675 Skeena Drive, Kamloops, BC, V2E 2M9, Canada

ABSTRACT

The Flammulated Owl (*Otus flammeolus*) Habitat Model for the Merritt and Kamloops forest districts was validated in these 2 British Columbia forest districts to test its ability to predict habitat variables. Comparative analysis of the model and field data indicated that the model accurately predicted leading tree species, aspect, slope, and forest age class for the Merritt Forest District alone and for both districts combined. The model tended to predict habitat suitability levels based on stand structure characteristics, grazing intensity and levels of western spruce budworm (*Choristoneura occidentalis*) activity observed in habitat suitability polygons in the Merritt Forest District. Predicted habitat suitability levels showed characteristics similar to distinct habitat types observed on Wheeler Mountain that are strongly suspected to indicate varying productivity levels; long-term trend data suggest moderate to high productivity is likely associated with particular forest cover and terrain features. Nest-site searches are recommended for subsequent validation of habitat utilization. Habitat features associated with productivity are critical to long-term management and must be the basis for habitat modelling.

Key words: flammulated owl, habitat modelling, *Otus flammeolous*.

A habitat suitability model was developed for the flammulated owl (*Otus flammeolus*) in 1998 to predict potential nesting habitat in the Kamloops and Merritt forest districts of British Columbia (Christie and van Woudenberg 1998). The model was created as a management tool to help with monitoring this Blue-listed species in south-central British Columbia, the northern limit of its range (McCallum 1994). Nest-site surveys would be too costly and time-consuming for locating nesting habitat throughout the 2 forest districts, and nesting habitat cannot be extrapolated directly from auditory census results (McCallum 1994, van Woudenberg 1999).

The model was based primarily on data from known nesting habitat on Wheeler Mountain, near Kamloops, B.C., using relevant forest cover and terrain variables (Christie and van Woudenberg 1997). Breeding habitat data for the Kamloops and Merritt forest districts were supplemented by data from Penticton (Cannings and Cannings 1982).

Habitat suitability model validation requires field-truthing at 2 levels. First, testing the model's ability to predict habitat features accurately in the field based on terrain and forest cover variables. Second, testing the model's ability to predict flammulated owl habitat use and levels of use predicted by habitat quality assignments.

Model validation began in the 1998 spring field season using vegetation plots to sample habitat features. The sampling design was a stratified subset of polygons from each habitat quality assignment, with representative polygons from 3 classes of habitat and 2 classes of nonhabitat from both the Merritt and Kamloops forest districts. Three plots were sampled per polygon, so that their compilation permitted the polygon to be used as the sampling unit.

STUDY AREA

The Merritt and Kamloops forest districts are located in south-central British Columbia in the Thompson-Okanagan Region and Kamloops Forest Region (Fig. 1). The districts range between Very Dry and Transition climatic regions (Lloyd et al. 1990). The habitat model restricted the study area in the districts to the Ponderosa Pine (PP) and Interior Douglas-Fir (IDF) biogeoclimatic zones. The area is characterized by gentle, rolling terrain, with steep areas that ascend to plateaus. The study area is primarily within the IDF zone, which occupies valley side-slopes. PP and Bunchgrass (BG) zones occur below, on lower slopes and in valley bottoms, while Montane Spruce (MS) and Englemann Spruce–Subalpine Fir (ESSF) occur on the plateaus above the elevation of the study area. Where the terrain is steeper (for example, on the western edge of the Merritt Forest District), the

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

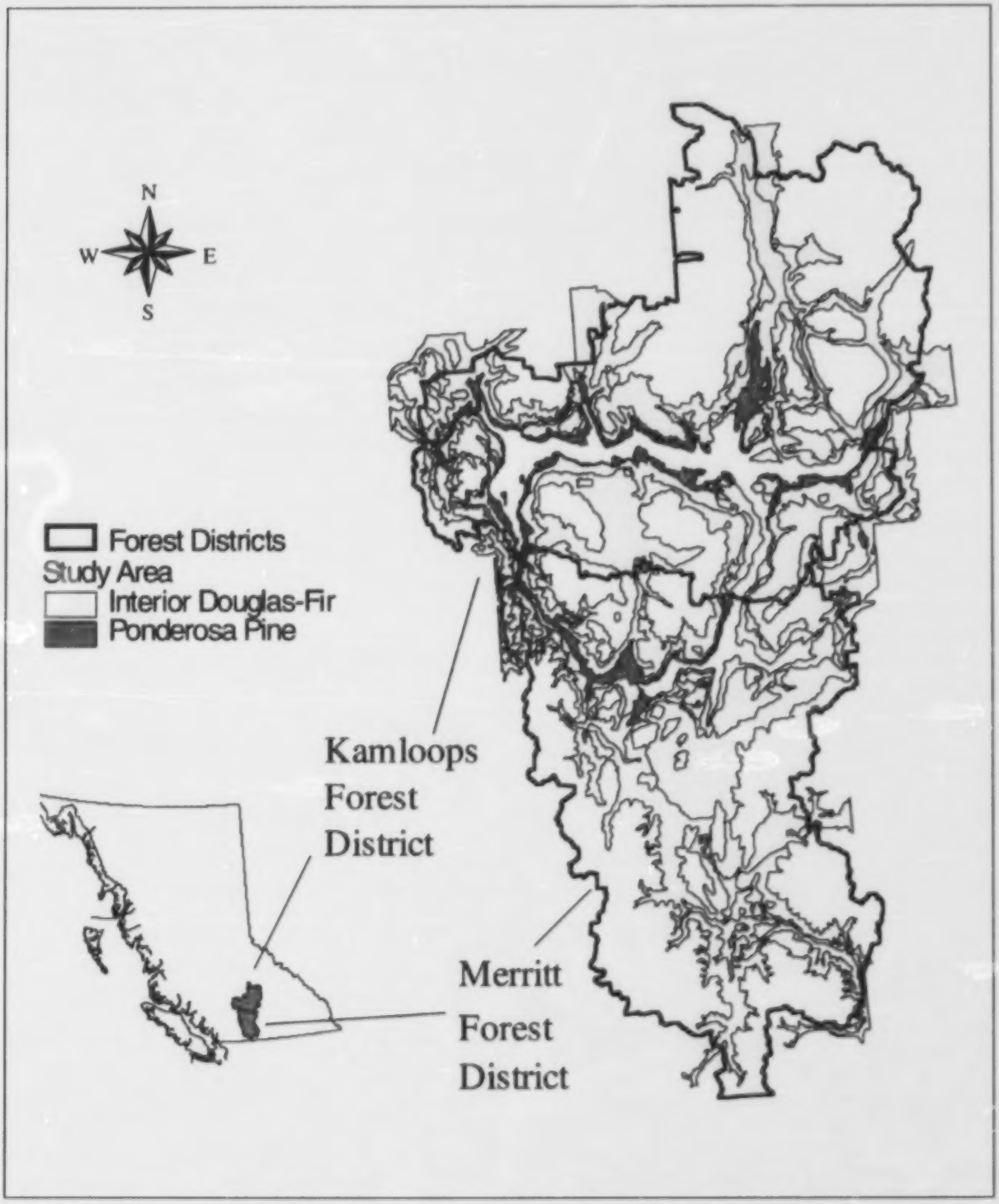

Figure 1. Thompson-Okanagan Region and Kamloops Forest Region study area.

Table 1. Variables used to derive the model.

Terrain variables	Forest cover variables	Biogeoclimatic variables
slope	age	variant
aspect	diameter at breast height	
	crown closure	
	leading species	
	subdominant species	
	activity[a]	

[a] Includes any past forest management activity as described by forest cover map variables (logging, planting, commercial thinning, etc.) qualified by year of that activity.

PP zone and mid-slope benches, where flammulated owl habitat tends to occur, are absent and were therefore excluded from the study area.

METHODS

Sampling Design

A stratified sampling design was used to randomly select polygons of each habitat suitability class (high, moderate, low, questionable or nonhabitat, and excluded habitat). The polygon was the sampling unit, sampled using 3 circular, randomly assigned plots no closer together than 150 m as per *Procedures for Environmental Monitoring in Range and Wildlife Habitat Management* (Habitat Monitoring Committee 1990).

The model variables (Table 1) and habitat descriptors were sampled in the plots (Table 2).

Plot locations were digitally recorded using a Trimble GPS (global positioning system) unit (Trimble Navigation Ltd. 1996). GPS location records ensured accuracy of plot placement within a polygon and provided a linkage between the field and model databases for statistical analysis.

Statistical Analysis of Model Data Accuracy

Two types of data from 3 data sources were used to create the model. Terrain variables included slope and aspect derived from a Digital Elevation Model that was created using the TRIM (Terrain Resource Information Management) database. The terrain variables and biogeoclimatic variants comprised the capability criteria of the habitat model. The forest cover variables were derived from the British Columbia Ministry of Forests forest cover database (FIP files) and concluded the suitability criteria of the model. Forest cover variables included leading species, diameter at breast height (DBH), crown closure, and age class.

To compare the field and model data, the habitat polygon was used as the sampling unit. Polygon level estimates were derived by averaging the 3 vegetation plots. Variables were compiled according to the VRI (Vegetation Resource Inventory) standards and procedures, which do not differ from those of the forest cover variables used for the model.

Table 2. Variables sampled in vegetation plots (12.6 m radius).

Terrain variables	Forest cover variables	INCOSADA variables[a]	Nonmodel variables
slope	age	thicket pattern	site series
aspect	diameter at breast height	grass pattern	slope position
	crown closure	shrub pattern	fire[c]
	leading species	snag class	
	subdominant species		
	activity[b]		

[a] INCOSADA = Inventory Corporate Spatial Attribute Database

[b] Includes any past forest management activity as described by forest cover map variables (logging, planting, commercial thinning, etc.) qualified by year of that activity.

[c] Recorded whether or not fire scars were evident.

The Shapiro-Wilk test ($\alpha = 0.05$) showed that data for each model variable had a nonparametric distribution. A sign test was selected for slope, age, and crown closure class variables since these were ranked, or ordinal, data. DBH values were continuous data, and therefore the Wilcoxon Signed Rank Test was selected to test this variable. For categorical variables (aspect, leading species 1 and 2, and activity), Fisher's Exact Test was used; this test does not require a normal distribution or a large sample size. There was a fundamental difference in the interpretation of the null hypothesis for each test. The sign test assumed there was no difference between the model and field variable; the Wilcoxon test was a test for independence, indicating the model and field variables were not independent in a significant test result (see project report: van Woudenberg et al. 1998).

Two variables could not be evaluated because they were not measured in the field. The Kamloops field data set did not include biogeoclimatic validation; and the activity year was either very difficult or not possible to be sampled in the field.

RESULTS AND DISCUSSION

The Flammulated Owl Breeding Habitat Prediction Model for the Merritt and Kamloops Forest Districts was validated for its ability to accurately predict forest and terrain variables relative to habitat features in the field. The success of the model to predict high-, moderate-, and low-suitability breeding habitat was not field-tested, since nest-site surveys were not conducted. Auditory surveys alone cannot be used to extrapolate nesting activity (McCallum 1994; B. Linkhart, University of Colorado, 1998, pers. comm.; van Woudenberg and Christie 1997). Owls calling spontaneously early in May can be indicative of breeding or nesting habitat, however,

owls calling later in the season may indicate suboptimal or nonbreeding habitat (van Woudenberg 1999).

A total of 23 polygons and 74 plots in 5 habitat suitability classes were sampled (Fig. 2).

Slope and aspect variables were accurately predicted by the model in the field both for the combined data and for each forest district individually. For the combined data from the two forest districts, and for the Merritt data alone, lead tree species was accurately predicted in the field; for the combined data, secondary species was accurate as well. Only the Merritt field data corresponded with the model's age variable. Data for both Kamloops Forest District alone and the combined districts failed to show a predictive relationship between the model and field data for the age variable. The contingency table created as part of the analysis suggested that there was some association between the model and field age variables for the Kamloops data, suggesting that the association was likely skewed (bias resulting from consistent sampling error), thereby resulting in a significant test result. Perhaps an age class of tree other than the dominant layer was consistently selected for increment boring to measure age in the Kamloops Forest District plots; for example, the average tree may have been selected rather than the dominant tree. Therefore, because of the distribution of the Kamloops data, age was most likely predicted accurately by the model and sampled incorrectly, although consistently, in the Kamloops Forest District.

Three variables were not well predicted by the model: crown closure, activity, and DBH. Crown closure is a subjective measurement in the field and can be somewhat subjective in the original photo-interpretation for forest cover inventory. Kamloops crown closure data was accurately predicted in the field; data for Merritt and the combined districts were not. For the activity variable there was a significant difference between the model and the field data. This may be due to the compilation criteria used to select the representative polygon value from the 3 documented plots. Field observations may have determined that logging or other recent management activity predicted for a site may not have been the greatest impact on the site. In addition, the time lag between the forest cover inventory database update and the current condition of the stand in the field may have led to a different interpretation in the field, particularly where juvenile spacing may have occurred.

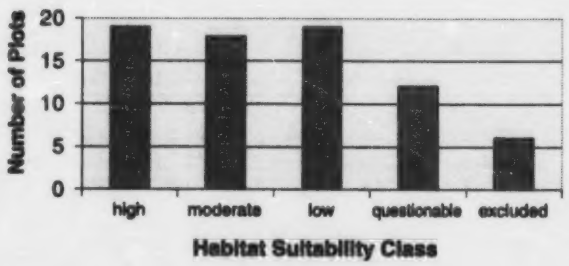

Figure 2. Vegetation sampling effect (n=74).

Finally, DBH was not accurately predicted by the model. This may be due to high variability in the forest cover inventory database resulting from photo-interpretation, or variability in the field data collection. Several trees were measured in the field and, according to the VRI standards, DBH should be the average for all live trees >2 cm diameter. The sampling method measured all trees >10 cm diameter. In plots with extensive thickets where high numbers of trees would be <10 cm but >2 cm, the DBH sampling method would have biased the data. DBH should have been measured for each canopy layer in order to discriminate the dominant layer and quantify stand structure (van Woudenberg et al. submitted).

The variable describing the biogeoclimatic ecosystem classification (BEC) variant could not be validated because it was not collected in the Kamloops Forest District. The Merritt Forest District sample on its own provided too limited a sample size to be statistically tested. The BEC variant is important data for habitat prediction because different forest age classes in different variants will provide critical habitat features (D. Low, B.C. Ministry of Environment, Lands and Parks, pers. comm.). As well, accurate habitat quality predictions in the field using BEC information, including site series, is very useful to forest licensees for operational planning.

The Flammulated Owl Breeding Habitat Prediction Model cannot be expressed as a single mathematical equation, such as a quadratic equation, because there is no 1 relationship used in its derivation. Multiple combinations of variables and subgroups of variables determine a single habitat quality assignment, and variable subgroups and/or combinations differ for different habitat quality assignments.

Habitat Descriptions of Quality Assignments for the Merritt Forest District

Overall, the stand and understory structure of the high- and moderate-suitability habitat class polygons tended to conform to the multi-aged stand structure documented in habitat studies from south-central British Columbia (van Woudenberg 1999). Stem densities measured in plots suggested that higher suitability habitat classes tended to have more open stands than nonhabitat and excluded habitat. The forest cover variables sampled suggest high-suitability stands had greater diversity of tree species composition relative to moderate- and low-suitability habitat types (van Woudenberg et al. 1998). Polygons encompassing nonhabitat or questionable habitat tended to have higher variability in tree species composition, which was expected since they were less well defined by stand parameters.

The abundance and species of vegetation sampled in plots in the high-suitability habitat type, as well as site series results (Fig. 3), suggest that several plots were placed in riparian areas as a result of random plot placement. Random plot placement should have been restricted to upland habitats, since flammulated owls are intolerant of humidity and prefer the dry, open upland forest structure (McCallum 1994). A potentially disproportionate number of plots placed in riparian areas of high-suitability habitat polygons may also have accounted for a lower than expected detectability of fire evidence in high-suitability habitats (van Woudenberg et al. 1998).

High-suitability habitat types had shrubs of moderate height with the highest percent cover and moderate amounts of grass cover (van Woudenberg et al. 1998). Moderate-suitability habitat types tended to have taller shrubs with less cover than high-suitability sites, but had a more developed grass layer. Low-suitability sites tended to have less developed shrub and grass layers, indicated by height and cover. The results are indicative of the xeric (high-suitability) and mesic (moderate-suitability) habitat types described for flammulated owls (van Woudenberg et al. submitted).

Understory development can be an important component of flammulated owl foraging habitat, due to enhanced availability of large-bodied insects (van Woudenberg 1999). Xeric sites associated with nests tended to have more shrub development, while nest sites in mesic habitat types had more grass development, indicative of potentially different foraging strategies. Grazing may impact forage habitat by reducing ground cover for insects. The results suggest that lower quality habitat may have had greater impacts from grazing. Grazing occurred in moderate amounts in all areas sampled, but was highest and most recent in low-suitability and non-habitat classes.

The incidence of site series (Fig. 3) also indicate the distinction between xeric (site series 3 and 4) and mesic (site series 5 and 6) habitat types that flammulated owls will use (van Woudenberg 1992). The site series results tend to agree with results from Wheeler Mountain regarding habitat types and suspected productivity, where habitats of higher suitability tend to be drier. Flammulated owls prefer dry, hot microclimates, where large insects may be either more abundant or more active, particularly at night. However, productivity may differ in the same habitat types in more southerly sites of the Merritt Forest District. Nesting and fledging activity data are necessary to determine if drier site series are more productive in the Merritt Forest District.

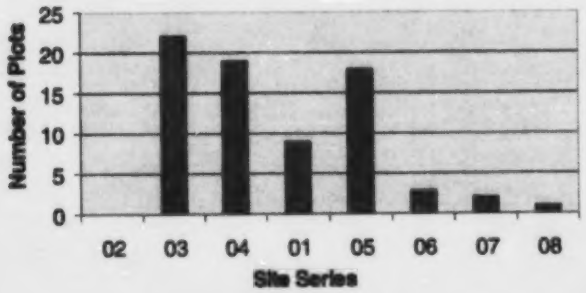

Figure 3. Incidence of reported site series in plots for the Merritt Forest District.

Western spruce budworm (*Choristoneura occidentalis*) activity was moderate throughout the study area in all habitats sampled. However, current and high levels were observed in high- and low-suitability classes. Adult flammulated owls have been observed to glean budworm larvae from Douglas-fir thickets (van Woudenberg and Christie unpubl. data). During the 1998 owl survey the budworm cycle was at its peak. Flammulated owls may opportunistically track budworm outbreaks in Douglas-fir–ponderosa pine habitat (van Woudenberg 1992).

RECOMMENDATIONS

Nest surveys are highly recommended for subsequent field validation of the Flammulated Owl Breeding Habitat Model. Auditory surveys, which were conducted during the 1998 field season in habitat classes, are recommended for stratifying areas for site selection for nest-site surveys in the next field season. Stratification should use 2 criteria: timing of the field season when birds were heard calling; and numbers of birds detected along transects (van Woudenberg and Christie 1997). A transect methodology is recommended, similar to that used for pest (bark beetle) management surveys. Auditory census results should never be extrapolated to assume breeding habitat (McCallum 1994). Even in early May, detected birds may only be passing through an area en route to suitable nesting habitat. Productivity information (nest surveys) is critical for verifying model habitat quality assignments.

Biogeoclimatic information, including site series, should be collected for model habitat variable validation to facilitate extension of project results to forest managers, including licensees and Forest Service personnel.

ACKNOWLEDGEMENTS

The project was funded by Forest Renewal British Columbia (FRBC) and administered by Weyerhaeuser Canada Ltd. The British Columbia Ministry of Environment, Lands and Parks was the technical partner. Analysis of the Kamloops Forest District data was not funded and was provided by Cascadia Natural Resource Consultants; the Kamloops data were collected as part of another FRBC project. D. Falat contributed to all aspects of this project and is gratefully acknowledged.

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Modeling Carnivore Translocations: An Example Using Canada Lynx

T.D. Steury
Department of Fish and Wildlife Resources
University of Idaho, Moscow, ID 83844 USA
steu4718@uidaho.edu

D.L. Murray
Powell Ranger Station, Clearwater National Forest
Powell Ranger District, Lolo, ID 59847 USA

ABSTRACT

The release of wild carnivores to native habitat has become an increasingly important method of re-establishing viable populations, but few guidelines have emerged to ensure that release success is maximized. Lynx may be an ideal carnivore for studying factors related to success of translocation, as relationships between lynx demographic parameters and densities of their main prey, snowshoe hares, are both closely associated and well studied. We mathematically modeled lynx reintroduction in order to predict which factors could play roles in determining release success. We examined deterministic, stochastic, and cyclic models of lynx-hare interactions, with parameter estimates derived from the literature. The functional relationship between various demographic attributes of lynx (productivity and survival of different-aged cohorts) and changing hare density was best represented by the logistic curve. In our modeled system, mean prey density and the year-to-year variability in prey density appeared to play the largest roles in determining success of translocation. We found that for most model iterations, a minimum density of approximately 1.3 hares/ha was necessary for a successful release, implying that many states in the contiguous United States may lack hare densities capable of supporting reintroduced lynx populations. We also found that both number of released animals, and number of releases positively affected translocation success. In addition, we found that in our modeled system, human-caused mortality, such as illegal trapping or collisions with vehicles, may significantly decrease probability of successful translocation. We conclude that, for carnivore species whose numbers are closely associated to prey abundance, the population ecology of prey in the release area should be determined prior to translocation. The appropriate number of animals to be released should also be determined, such that release success is maximized while minimizing cost. Finally, the most important consideration when preparing for carnivore translocation may be the potential human disturbance of translocated populations.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Quantitative Tools for the Monitoring of Threatened Species: A Marbled Murrelet Case Study

John G. Boulanger
Integrated Ecological Research
924 Innes Street, Nelson, BC, V1L 5T2, Canada
boulange@ecological.bc.ca

ABSTRACT

Recent developments in quantitative methods can potentially improve the statistical rigor of monitoring programs. Matrix modelling and power analysis are valuable tools in the design of monitoring programs even for species in which sparse data exist. Newer distance and mark-recapture methods can improve the quality of estimates when compared with count-based methods. This paper presents a case study of the marbled murrelet (*Brachyramphus marmoratus*) to illustrate the application of these methods. Two possible murrelet life history scenarios that represent the extremes of plausible murrelet parameter values, given the limitations of current knowledge, are examined to illustrate matrix modelling and power analysis. Sensitivity analysis suggests that regardless of life history scenario, murrelet populations are more sensitive to changes in adult survival, and less sensitive to juvenile survival and productivity. Power analyses of population trajectories generated by the matrix model suggest that only large reductions in individual population parameters would be detected in 10 years by at-sea census techniques. Results of these analyses have helped to prioritize strategies for monitoring and research of murrelet populations. The results of recent studies are used to illustrate the applicability of mark–recapture and distance methods to marbled murrelet populations. Estimation of survival rates for marbled murrelets in British Columbia using mark–recapture methods is providing new insight into the life history of the murrelet. Distance methods improved power of at-sea surveys of murrelet populations in California. Comprehensive power analysis (MONITOR), mark–recapture (MARK), and distance (DISTANCE) software programs are currently available free from specialized Web sites.

Key words: distance sampling, mark–recapture, matrix models, population monitoring, power analysis, uncertainty.

Population monitoring programs involve the complex task of detecting demographic trends, which are the result of the interaction among species demography, environmental perturbations, and stochastic effects. This paper focuses on how quantitative methods can potentially improve the statistical rigor of monitoring programs.

Matrix modelling and power analysis are valuable tools in the design of monitoring programs even for species in which sparse data exist. Many recent studies have used matrix model methods to evaluate conservation measures and forecast population trends for rare species (Crouse et al. 1987, Caswell 1989). These mathematical studies have mainly been conducted for species in which substantial research efforts have produced precise parameter estimates. However, for many species, not enough data currently exist to allow precise parameter estimates; therefore, little effort has been made to understand the species population dynamics using the rigorous framework of a population model. Sometimes the lack of understanding regarding the effect of specific parameters on overall population dynamics has led to unfocused research and conservation efforts.

A case study of marbled murrelets is used to illustrate the basic steps in the matrix model and power analysis approach to monitoring study design. This paper cannot fully document the modelling process; refer to Boulanger et al. (in press) for a thorough discussion of this case study. Power analysis of monitoring data may reveal a low probability to detect a population change and discussion is provided on alternative methods to boost power.

METHODS

Matrix Models

Various matrix and population viability analysis (PVA) models are available for use in species management and monitoring design. These models vary in complexity from simple matrix models to spatially explicit metapopulation models. This paper focuses on the use of simple matrix models and power analysis to prioritize monitoring objectives. For this

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paper, matrix models are used to explore monitoring strategies but cannot be used to determine absolute rates of population change given the high uncertainty in model parameters. Readers should consult Beissinger and Westphal (1998) for the advantages and pitfalls of PVA methods.

Step 1: Conceptualisation of Life History Parameters and Formulation of Matrix Model

General life history parameters and scenarios for marbled murrelets were initially determined by discussion between field biologists and quantitative ecologists and review of existing literature. From this process a simple life history model for marbled murrelets was formulated (Fig. 1). Each node in the diagram represents a specific stage, which is controlled by a unique set of life history parameters. In this model the population is censused in the fall after the breeding season. The survival of young of the year for the 2 years is parameterized by juvenile survival (S_j) and after that parameterized by adult survival (S_a). The breeding age birds (assuming an age of first breeding of 2 years) attempt to breed each spring and their success is parameterized by productivity (P). The numerical relationships in the life history diagram can then be easily translated into matrix format.

The projections from the resulting matrix model follow the model of exponential population growth (or decline) as defined by the equation:

$$N_t = N_0 e^{rt}$$

In this equation, N_0 is the initial population number and N_t is the value at a give time t. The dominant eigenvalue of the matrix is symbolized by λ and is equal to e^r. If the population is stable then λ must equal 1. If the population is decreasing, λ is negative, and if the population is increasing, λ is positive. Therefore, by calculating λ from the matrix the future population trajectory for a set of population parameters can be determined (Caswell 1989). Note that the basic matrix calculations can be programmed on an Excel© or similar spreadsheet package whereas more complex matrix calculations require specialized programs such as Matlab© or O-Matrix©.

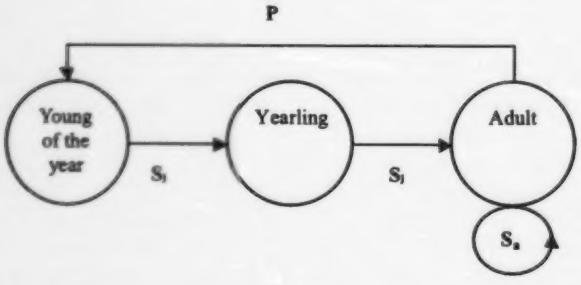

Figure 1. A life cycle diagram for the marbled murrelet for age of first breeding at 2 years. This life history diagram pertains to a population in which a post-breeding census is conducted. P = productivity, S_a = adult survival, S_j = juvenile survival from first fall census to 2 years of age.

Some assumptions apply to this matrix model. First, the model considers that only females are included, breeding age females always find mates, and the population has an even sex ratio. Second, the population is censused in the fall after breeding. Third, the population is at equilibrium and always has a stable age structure. Fourth, the model is linear with no density-dependent effects. Finally, we assume that senescence is solely determined by adult survival values. The matrix model can be easily altered to generate precise predictions which can be used to test these assumptions with field data.

Step 2: Estimation of Life History Parameters and Potential Life History Scenarios

The second step in the modelling process involved assigning likely ranges of life history parameters based on sparse current knowledge of murrelets. Inference from comparison of similar alcid species by Beissinger (1995) gave approximate ranges of survival values for murrelets. Beissinger (1995) estimated that murrelets in their first year of life survive at 70% of adult survival and murrelets in their second year of life survive at 88% of adult survival. To keep the model simple, pre-breeding survival was compressed into 1 parameter by taking the geometric mean of 0.88 and 0.70. Juvenile survival was then calculated by multiplying this value (0.78) by the adult survival for each simulation. Data from observed nests and juvenile adult ratios gave approximate productivity values (Beissinger 1995, Nelson and Hamer 1995, Boulanger et al. in press).

A wide range of life history parameter values were plausible given the uncertainty in the life history of the murrelet. One initial method to formulate likely combinations of life history parameters was to determine which combinations result in a stable population. This was done by iteratively solving the matrix for parameter combinations which resulted in $\lambda = 1$ (Fig. 2).

To account for the large degree of uncertainty in parameter values Boulanger et al. (in press) focused on 2 life history scenarios with parameter values that maintain a stable population (Fig. 2). In the "high productivity" scenario, productivity could possibly be higher than that reflected by juvenile/adult ratios. With this scenario, adult survival falls within the range suggested from allometric comparisons with other alcid species and recent mark–recapture survival rate estimates . The second "high survival" scenario implies that murrelets have higher adult survival than suggested by their body size. This scenario is most plausible if we assume that current juvenile/adult ratios in Beissinger (1995) reflect true productivity.

This "extreme scenario" approach is a simple way to account for uncertainty in life history parameters. The main rationale behind this approach is that the true life history

parameters for murrelets fall somewhere between the 2 extreme values. Modelling the 2 scenarios allows a test of model robustness to uncertainty in life history parameter values.

Step 3: Sensitivity Analysis

Sensitivity analysis was used to investigate the relative importance of each life history parameter in maintaining murrelet population given the range of parameter values previously listed. In each analysis, the change in λ was recorded when each life history parameter was proportionally decreased while the others were held constant (Fig. 3).

Results from the sensitivity analysis suggest that murrelet populations are most sensitive to changes in adult survival, followed by juvenile survival and productivity (Fig. 3). Marbled murrelets may have evolved a life history strategy that tolerates large fluctuations in yearly productivity that has been observed in other seabird species. This strategy would partially explain low model sensitivity to productivity. Given this, Boulanger et al. (in press) recommended that monitoring should emphasize detailing variances in productivity and estimating survival using methods similar to Kuletz and Kendall (1998) and Lougheed et al. (1998).

Step 4: Power Analysis

Power analysis of population trajectories generated by the matrix model allows a testing of how well current monitoring can detect hypothetical changes in population. Boulanger et al. (in press) used the methods of Gerrodette (1987) in which the main input variable for analysis is coefficient of variation, which can be measured from replicated survey data. Coefficients of variation from replicated strip transect surveys conducted by Strong et al. (1995) were used for this analysis. This simplistic power analysis model was useful for approximate power calculations (Link and Hatfield 1990). More sophisticated and robust Monte Carlo simulation-based power analysis models in program MONITOR (Gibbs 1995) are available for free on the World Wide Web (Table 1).

The power analysis of Boulanger et al. (in press) suggests that reductions in population due to adult survival decrease are much more likely to be detected than changes due to juvenile survival or productivity if strip transects methods with precision levels of Strong et al. (1995) are used in surveys (Fig. 4). Murrelet researchers may have to use more direct measures of productivity such as the methods discussed by Kuletz and Kendall (1998). In addition, factors affecting adult survival such as fishing net mortality should be considered when large murrelet declines are detected in short periods of time Boulanger et al. (in press).

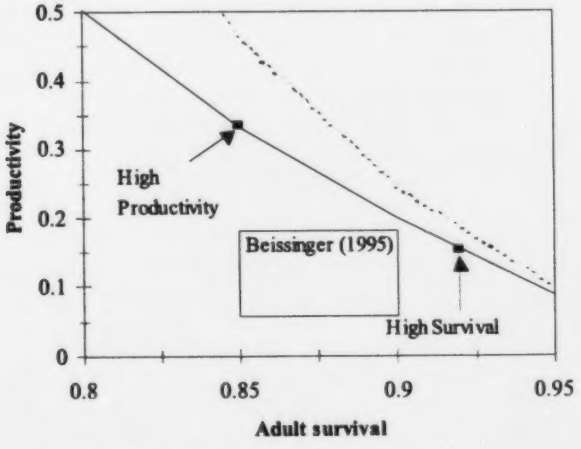

Figure 2. Combinations of adult survival and productivity resulting in a stable population conducted for ages of first breeding of 2 (solid line) and 4 years (dashed line). Labelled points are the parameter values in which the analysis of Boulanger et al. (in press) was conducted. Note that juvenile survival is assumed to be 78% of adult survival in this model and therefore is not shown on axis. The survival and productivity values proposed by Beissinger (1995) for murrelet populations in the Unites States are bounded by the labelled box.

ALTERNATIVE MONITORING STRATEGIES

This section discusses alternative monitoring strategies that can increase the statistical power and rigor of monitoring programs.

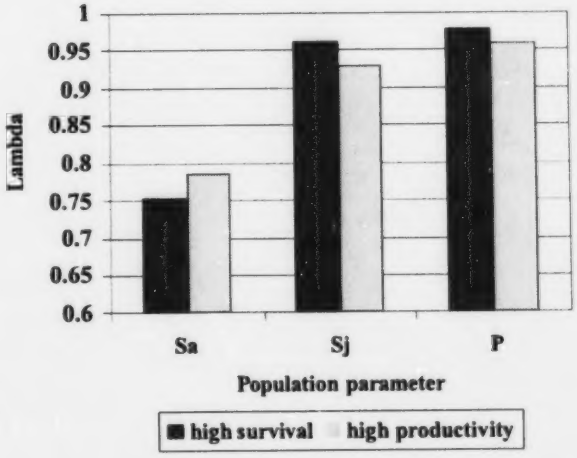

Figure 3. Results of sensitivity analysis (individual parameters reduced by 30%) of high survival and high productivity scenarios. P = productivity, S_a = adult survival, S_j = juvenile survival.

Table 1. Web site addresses for program downloads.

Program	Web address
MARK	<http://www.cnr.colostate.edu/~gwhite/software.html>
DISTANCE	<http://www.ruwpa.st-and.ac.uk/distance/>
MONITOR	<http://www.im.nbs.gov/>
Related programs	<http://www.ecological.bc.ca/links>

Distance Methods

Often the sampling design for count-based indices is either the point count or strip transect in which all the animals within a specified distance from the survey line or centre are counted. A fundamental assumption of strip transects is that all animals, or a constant proportion of animals, are counted within the survey strip.

When compared with strip transects, the only additional measure needed for the use of distance methods is the distance of individuals (or clusters of individuals) from the survey centre. Data can be measured by distance categories or groupings when it is difficult to get exact distances. Therefore, distance methods can often be used in field situations with little increase in effort (Buckland et al. 1993).

With distance methods, all animals sighted (regardless of distance from the survey centre) are counted and the sightability of animals is estimated, therefore allowing an actual estimate of population density (and associated variance). This method is unlike strip transects which usually underestimate density (due to violation of the assumption of all animals being sighted within the strip) (Burnham and Anderson 1984). In addition, a more robust estimate of survey precision is possible with distance methods. Finally, the seldom tested assumption *that index counts and true abundance are* linearly related is less likely to be violated with distance methods (Burnham and Anderson 1984, Buckland et al. 1993). All of these attributes may contribute to the increase in power and statistical validity with distance methods.

A common reason for not attempting to use distance methods is sample size constraints. Buckland et al (1993) state that most distance models need at least 60 observations to ensure adequate precision. However, if the sightability of animals is high and factors affecting detection rate are not complex (due to proper sampling design), adequate results can be obtained with less than 60 observations (pers. comm., S. Buckland, Mathematical Institute, Univ. of St Andrews, Scotland). Recently, a comprehensive, Windows®-based distance calculation (DISTANCE) has become available and can be downloaded free from the World Wide Web (Table 1).

The increase in power with distance methods with marbled murrelet monitoring is exemplified by the results of Becker et al. (1997) in which the power of strip transects and distance methods (line transects) was compared for marbled murrelet populations off the coast of California. In short, Becker et al. (1997) showed 2.3 times increase (0.24–0.57) in power between line and strip transects. The line transect method always resulted in higher counts of murrelets which most likely contributed to increase in statistical power.

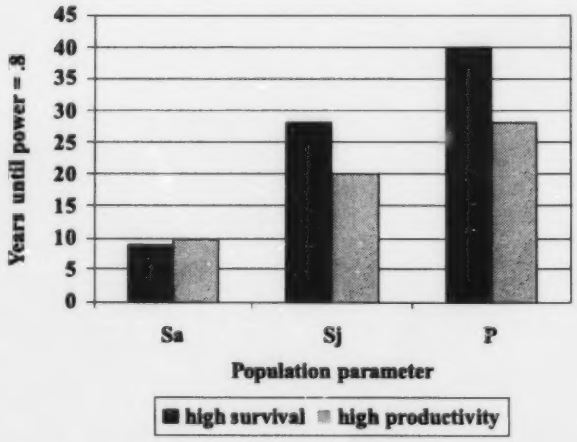

Figure 4. Results of power analysis, based on line transect surveys documented by Strong et al. (1995) for high survival and high productivity scenarios. The *y*-axis represents the number of years it would take to statistically detect a population decrease resulting from a percentage reduction in a given parameter. P = productivity, S_a = adult survival, S_j = juvenile survival.

Mark–Recapture Methods

Mark–recapture methods are similar to distance methods in that the capture probability of animals is estimated which allows a more statistically rigorous estimate of population trend. Historically, mark–recapture methods such as the Jolly-Seber model displayed marginal precision due to the large potential number of parameters being estimated as part of survival rate estimates. Recently, constrained versions of the Jolly-Seber model, and advanced models that include survey covariates (such as effort, weather factors, and biological attributes), have allowed estimates with increased precision and reduced bias (Labreton 1992, White 1998).

The choice of whether to use mark–recapture models is often determined by the viability of obtaining adequate samples of animals in the field given costs and other logistical constraints. In addition, the use of Jolly-Seber models requires a commitment of at least 3 years to obtain survival rate estimates. However, obtaining a viable population trend arguably requires at least 3 years of data and therefore the constraint of time commitment is universal to all

monitoring methods (Gibbs 1995). Simulation modules incorporated in newer mark–recapture programs such as MARK (White 1998) and POPAN (Arnason et al. 1998) allow calculation of sample sizes from preliminary survey data (Table 1).

Capture–recapture techniques with marbled murrelets are exemplified by a study in Desolation Sound that uses newly developed mist net and dip net capture techniques. An adult survival rate was estimated at 0.845 (95% confidence limits 0.47 to 0.94) (Lougheed et al. 1998). This estimate is based on 4 sampling sessions (conducted yearly from 1994 to 1997); therefore, precision is low due to low numbers of recaptured birds. Future estimates from this project should provide critical information regarding life history parameter values of marbled murrelets as the sample size of marked birds and corresponding precision of estimates increase (Lougheed et al. 1998).

Use Power Analysis for Count Indices

If counts methods are used then the survey effort should be fine-tuned using specialized power analysis packages. For example, program MONITOR allows the user to vary the number of surveys per year and other survey parameters to determine the optimal survey effort (Gibbs 1995). Note that the user inputs hypothetical rates of population change, and MONITOR does not require knowledge of life history parameters or the use of matrix models. This program is available for free on the World Wide Web (Table 1).

When possible, monitoring efforts should be targeted towards life history stages as determined by the results of matrix model sensitivity analysis. For example, productivity indices of (Kuletz and Kendall 1998) can provide valuable information regarding annual variation in productivity. Becker et al. (1997) provide a good example of the use of program MONITOR in formulating optimal sampling strategies.

CONCLUSION

This paper introduces a general method to evaluate the uncertainty in life history assumptions and formulate monitoring priorities for species at risk. In addition, this method allows evaluation of how the uncertainty in life history parameters relates to field monitoring through power analysis. If data are unavailable for the use of matrix models, researchers must still use power analysis to evaluate monitoring programs. In addition, consider improved monitoring methods such as distance methods, which involve fewer assumptions and many times exhibit higher statistical power than count-based indices. Monitoring populations is a complex, costly, and challenging task. The use of quantitative tools is inexpensive compared with field efforts; these tools can supply valuable inference towards optimal monitoring design as well as more statistically robust results.

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Use of Geographic Information System and Radar Counts to Delineate Suitable Habitat and Estimate Population of Nesting Marbled Murrelets of British Columbia's Central Coast

Monica H. Mather[1]
3165 Rock City Road, Nanaimo, BC, V9T 1T5, Canada
mmather@mail.island.net

Bernard K. Schroeder
British Columbia Conservation Foundation
202-1696 Frances Street, Vancouver, BC, V5L 1Z4, Canada

Trudy A. Chatwin
British Columbia Ministry of Environment, Lands and Parks
2080-A Labieux Road, Nanaimo, BC, V9T 6J9, Canada
Trudy.Chatwin@gems1.gov.bc.ca

ABSTRACT

Broad Ecosystem Unit (BEU) and forest cover at the 1:250,000 scale had been mapped to contribute information to the Central Coast Land and Coastal Resource Management Plan. We ranked BEU and forest cover polygons based on their suitability for marbled murrelet nesting habitat. During the 1998 breeding season, we conducted a reconnaissance level survey of 27 selected watersheds draining into the central Coast. We used high-frequency radar to count marbled murrelets as they entered the watersheds before dawn. Geographic information system (GIS) mapping enabled us to readily calculate both the size of the watersheds and the amount of suitable nesting habitat available (based on both the broad ecosystem and forest cover information). Based on the radar counts we calculated an index of the number of nesting marbled murrelets per hectare of suitable habitat. This index was used to rank the suitability of the watersheds for marbled murrelets and delineate broad areas to be further examined as marbled murrelet nesting habitat. Our estimates of numbers of marbled murrelets flying inland to the central Coast study area ranged from 5,343 to 42,359. We believe the estimate of 21,000 birds, based on using the index for BEUs 1 and 2 suitable habitat, is most accurate.

[1] Present address: 7017 Lavender Road, Lantzville, BC, V0R 2H0, Canada

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Assessing Cumulative Impacts of Alternative Land Use Scenarios On Breeding Bird Habitat

Brenda Dobson
Parks Canada, Jasper National Park
Box 10, Jasper, AB, T0E 1E0, Canada
brenda_dobson@pch.gc.ca

ABSTRACT

Increasing recognition of the importance of understanding, assessing, and mitigating cumulative effects has led national park managers to seek methods that can assist in land use decision-making and the environmental assessment process. The objective of this research is to develop a geographic information system (GIS)-based model to investigate the cumulative impacts of alternative land use scenarios on 2 ecological indicators. The model incorporates ecological data and habitat models for a suite of ecological indicators and human use data managed by ARC/INFO software. A grizzly bear cumulative effects model has been developed for Jasper National Park to assess the effects of human activities on grizzly bears and their habitat. This model is useful in exploring the implications of land uses at the landscape scale, but has limited ability to assess alternative development scenarios at fine scales. To address this concern, 2 ecological indicators, ecosite representation and breeding bird richness, are being developed based on data collected in the Banff and Jasper National Park Ecological Land Classification. The responses of these indicators will be compared for scenario development at a fine scale, namely, the high human use montane area around Jasper Townsite. The results of the application of the scenarios to the ecological indicators will be measures of breeding bird habitat effectiveness and ecosite representation. This poster focused on the breeding bird richness indicator. Habitat effectiveness is the comparison between the habitat and the disturbance components of the models and represents an area's ability to support the indicator. A first run of the model is complete with preliminary results presented. Sensitivity analyses and scientific review of the zones of influence surrounding land use features are in progress.

Key words: Alberta, birds, breeding habitat, cumulative effects assesament, ecological indicators, habitat effectiveness, Jasper National Park.

The primary focus for most of the development proposals in Jasper National Park is an area called the Three Valley Confluence which includes the townsite, outlying resorts, a ski hill, and a golf course.

The Jasper Town Plan and many major land use proposals have stimulated the development of a cumulative effects assessment (CEA) framework for the park.

The breeding bird habitat effectiveness model presented here is being developed to facilitate the use of breeding bird habitat quality as an CEA indicator. Other elements of the framework include grizzly bear habitat effectiveness, linkage zone analysis, ecosite representation, and socioeconomic indicators.

The objective of this study is to investigate the cumulative impacts of human use on breeding bird habitat through the use of a geographic information system (GIS)-based spatial model which uses existing ecological data and applies alternative land use scenarios.

METHODS

The Banff–Jasper ecological land classification was developed between 1974 and 1981. It is based on an integrated resource inventory mapped at 1:50,000. Seventeen hundred 500-m call-count transects were completed within Jasper and Banff National Parks. Several samples of each ecosite were required due to the large study area and probability that wildlife use would vary across the same ecosite type. A transect was located within a single ecosite, with multiple transects per ecosite type.

Bird richness was calculated for 125 ecosite types. Each bird species was rated against disturbance criteria based on the scientific literature to predict displacement from habitat within a zone of influence (ZOI) surrounding a human use feature (the preliminary breeding bird classification by category was provided to obtain input from conference participants).

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The coefficients of disturbance represent the percentage of breeding birds within a suite which are predicted to be displaced from the ZOI based on the disturbance criteria (Fig.1).

SAMPLE CALCULATION FOR BREEDING BIRD HABITAT EFFECTIVENESS

PT4 ecosite types have a *bird richness* of 50. The area of this PT4 ecosite is 60 ha. Given the richness and the area, the *habitat potential* for PT4 is 50 x 60 = 3,000.

The area of the resort within PT4 is 40 ha. The *zone of influence* (ZOI) designated for a resort is 100 m, and the total area of zone of influence within PT4 is 5 ha.

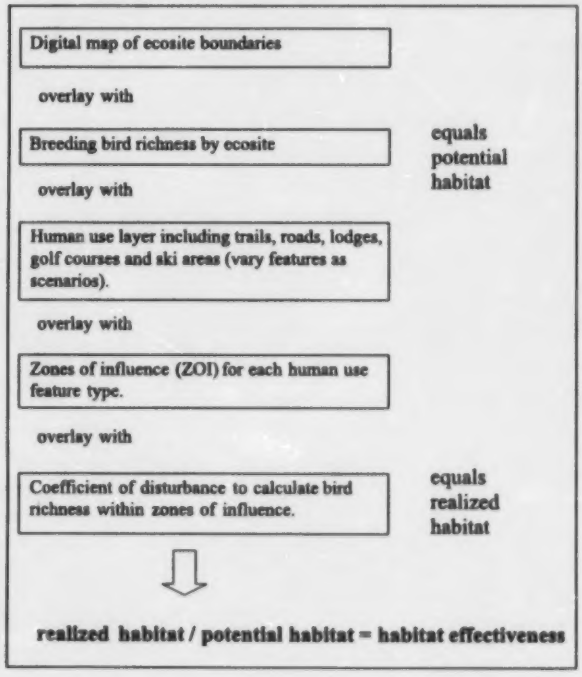
Digital map of ecosite boundaries
overlay with
Breeding bird richness by ecosite
equals potential habitat
overlay with
Human use layer including trails, roads, lodges, golf courses and ski areas (vary features as scenarios).
overlay with
Zones of influence (ZOI) for each human use feature type.
overlay with
Coefficient of disturbance to calculate bird richness within zones of influence.
equals realized habitat
realized habitat / potential habitat = habitat effectiveness

Figure 1. Data layer requirements for the breeding bird habitat effectiveness model for Jasper National Park, Alberta, Canada.

Of the suite of species occurring in PT4 ecosites, 10 out of 50 are predicted to be disturbed within 100 m of the resort. Therefore, the *coefficient of disturbance* within the ZOI is (50 – 10)/50 = 0.8.

The habitat value within the ZOI is 5 x 0.8 x 50 = 200.

This resort represents 40 ha of habitat with no value for breeding birds. The *realized habitat* is (60 – 40 – 5) x 50 + 200 = 950. Habitat effectiveness (realized habitat/potential habitat) is 950/3,000 = 0.32 or 32% *habitat effectiveness*.

STATUS OF THE MODEL

This model is in progress. A first run of the model is complete. Coefficients are being refined and land use scenarios are being developed. A sensitivity analysis for the model is planned.

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Identifying and Managing Fauna Sensitive to Forest Management: Examples From the Sicamous Creek and Opax Mountain Silvicultural Systems Sites

David J. Huggard
Centre for Applied Conservation Biology, Forest Sciences, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada
huggard@interchg.ubc.ca

Walt Klenner
Kamloops Forest Region, British Columbia Forest Service
515 Columbia Street, Kamloops, BC, V2C 2T7, Canada

Alan Vyse
Kamloops Forest Region, British Columbia Forest Service
515 Columbia Street, Kamloops, BC, V2C 2T7, Canada

ABSTRACT

An effective sensitive-species research and management program must include conservation of provincially rare species as well as: (1) knowledgeably managing relatively common species and habitat elements that are known to be sensitive to human activities, to prevent them from becoming rare; (2) maintaining throughout their range, species with critical functional roles ("keystones") in ecosystems; and (3) obtaining basic information on the many poorly known taxa. We illustrate how these 3 categories were used to identify priority taxa for study at the interdisciplinary silvicultural systems research sites at Sicamous Creek and Opax Mountain, and how the research may help keep species and ecosystems off our "at risk" lists. Examples discussed include species or habitat elements threatened or extirpated elsewhere (e.g., pine marten, spruce grouse, coarse woody debris), groups with known key roles (e.g., red-backed voles, snags), and little known taxa (e.g., beetles, grylloblattids, gastropods).

Key words: coarse woody debris, *Falcipennis canadensis*, *Martes americana*, Opax Mountain, research priorities, Sicamous Creek, silvicultural systems.

Biodiversity conservation in British Columbia is based primarily on managing for a range of diverse and representative habitats. Although this "coarse filter" approach may be sufficient to maintain 85–90% of species and the ecological processes that sustain them (Nature Conservancy 1982, Noss 1987), fine filter management is also required. A focus on preserving known species that are currently rare in British Columbia is a necessary, but not sufficient, "fine filter" strategy for preventing the long-term loss of biological diversity. Expanding the conservation of "species at risk" to include research into species that are not currently rare but that are sensitive to management will ultimately save money and species. Our emphasis is on forest-dwelling species, many of which are not officially "sensitive species," but are sensitive to forest management. Intensive forestry threatens some species through the loss of older seral stages and many other species through the loss of forest attributes, such as snags and coarse woody debris (CWD). A list of 1,487 threatened species in Sweden (Berg et al. 1994), a country with highly industrialized forestry, provides evidence of this risk. Additional species may be threatened by the shortening of early seral stages, or the elimination of special habitats created by natural disturbances (e.g., fire-killed wood used by many species of insects). For many species, the negative effects of intensive forestry may be mitigated by stand- and landscape-level management options, including alternative silvicultural systems (B.C. Ministry of Forests 1991), the retention of specific forest attributes in harvested areas (Thomas 1979), and long-term planning of harvest in time and space. However, current knowledge of most species in British Columbia is inadequate to support informed decisions on implementing these management options. Failure to obtain and apply this information will ultimately require expensive, contentious, and uncertain actions to retain or restore declining populations of many species.

The provincial Silvicultural Systems Research Program is aimed at experimentally testing operational alternatives to traditional silvicultural systems (i.e., large clearcuts, except in the Interior Douglas-fir zone where single-tree selection predominates) (B.C. Ministry of Forests 1991). The project sites

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provide opportunities for many researchers to study the effects of forest management alternatives on a wide range of ecosystem values; forest biodiversity is prominent among these. Based on our experience directing sporadically large faunal diversity research programs at 2 of these sites (Sicamous Creek and Opax Mountain), we suggest 3 criteria for determining priority species or higher taxa for applied research in developing a "preventative" sensitive species strategy. (See Vyse [1997] and Klenner and Vyse [1999] for an overview of the Sicamous Creek and Opax Mountain research.) Examples are all from our own research. However, we do not imply that our research is the perfect model (we would make many changes next time). More importantly, we believe that priorities for large research programs should be determined independently of specific interests of the researchers.

CRITERION 1. SPECIES EXTIRPATED, ENDANGERED, OR THREATENED ELSEWHERE (OR HABITAT ELEMENTS LOST)

Europe, Scandinavia, eastern North America, and the U.S. Pacific Northwest all offer opportunities to learn from forests similar to those of British Columbia, but with longer histories of management. In Scandinavia, an excellent knowledge of natural history has resulted in extensive reviews of endangered species and causes of endangerment (e.g., Berg et al. [1994]), while legal requirements have produced the same information in the United States (Flather et al. 1994). Many of the same vertebrate, vascular plant, and lichen species, or closely related ecological equivalents, are found in the boreal or temperate forests of North America and Eurasia, providing a direct opportunity to foresee what will likely become threatened in British Columbia. The longer-term effects of forestry, over several rotations, on habitat elements can also be measured directly in these areas (Angelstam in press).

Examples of priority studies at Sicamous Creek and Opax Mountain based on this criterion include martens (*Martes americana*), spruce grouse (*Falcipennis canadensis*), and coarse woody debris (CWD). Martens have been extirpated by forest harvesting in several U.S. states and are endangered in Newfoundland from intensive clearcutting (Thompson 1991); they are considered sensitive species in several U.S. national forests (MacFarlane 1994) and are used to guide management in Oregon and Washington (Thompson and Harestad 1994). Winter studies at Sicamous Creek suggested that marten can be maintained by using arrays of small patch cuts, with an emphasis on retaining advanced regeneration, patches of dense canopy, and wetter sites (Huggard in press a).

Spruce grouse have been extirpated in highly fragmented areas of eastern North America (Fritz 1979) and around the periphery of their range in central and western North America (Braun et al. 1994). The ecologically similar capercaillie (*Tetrao urogallus*) has been extirpated from areas of Europe and is threatened by industrial fragmentation throughout much of its range (Rolstad 1989), as is the Siberian spruce grouse (*Falcipennis falcipennis*) (Andreev 1990). Spruce grouse at Sicamous Creek were closely associated with knolls and areas near wetlands, difficult tree-growing sites that would be good locations for reserves. Despite a short negative edge effect beside openings, spruce grouse occurred more frequently in patch cut arrays than in uniform partial cuts (Huggard 1997).

Coarse woody debris (CWD) in Swedish forests was reduced to one-third the volumes in similar British Columbia forests after 1 rotation, and to less than one-twentieth the volume after 2 or more rotations (Angelstam in press). Over 550 species are endangered in Sweden by this loss of dead wood (Berg et al. 1994). Large-scale replicated CWD manipulations have been conducted in uncut and clearcut forest at Sicamous Creek and Opax Mountain to measure the effect of these possible future conditions on small mammals and amphibians (Craig et al. 1997, Ferguson 1999).

We encourage a review of all species threatened elsewhere in intensively managed boreal and temperate forests, especially those for which we have a high provincial responsibility.

CRITERION 2. "KEYSTONE" SPECIES OR HABITAT ELEMENTS

"Keystone" species were originally defined as those whose presence in a community was required to maintain the diversity of the community (Paine 1969). The term has since been broadened and weakened to mean any species that a researcher feels is important, and its utility questioned (Mills et al. 1993, Hurlbert 1997). Our use of "keystone" is more similar to Hurlbert's (1997) less familiar "functional importance," the sum of changes in productivity of all species that would occur if the species or habitat feature of interest was not in the system. Although impractical to measure directly, some examples are obvious, and the idea gave priority to several studies at Sicamous Creek and Opax Mountain.

Red-backed voles (*Clethrionomys gapperi*) in the forests of British Columbia's interior play numerous roles, from being the primary food for many carnivores and raptors (especially those found in mature or old forest) to indirectly affecting numerous species through their contributions to forest productivity and heterogeneity. By dispersing ectomycorhizzal and wood-decaying fungi, red-backed voles enhance young conifer growth and decay of dead wood. This decay contributes habitat for many invertebrates and is a part of forest nutrient cycling. Red-backed voles can also affect regeneration by feeding on some seedlings or eating conifer seeds, while the roots of adult subalpine fir trees can

be extensively damaged by vole feeding (Merler 1997). These feeding activities produce heterogeneity in the forest, and the latter contributes to numerous patches of snags, required by various species. Work at Sicamous Creek and Opax Mountain shows that red-backed voles can be maintained by uniform partial cutting, or arrays of small patch cuts, if shrub cover and at least moderate volumes of CWD are maintained (Klenner 1997, 1999).

Snags are an obvious keystone habitat element, required by many vertebrates, invertebrates, and lichens (Thomas 1979, Berg et al. 1994, Goward and Arsenault 1997), and a source of future CWD, but also a potential hazard to forest workers. Abundances and types of snags needed by woodpeckers, small cavity-nesters, and lichens are being studied at Sicamous Creek and Opax Mountain (Goward and Arsenault 1997, Klenner and Huggard 1997, Klenner and Huggard 1999). A project at Sicamous Creek identified snag classes with a low risk of falling (Huggard in press *b*), but incorporating snag retention into partial cut systems remains a challenge.

We recommend identifying several diverse keystone species and elements in forested systems. Ideally, their functional importance would be tested through manipulative experiments (Mills et al. 1993, Hurlbert 1997). Because of their important roles, research should be directed at maintaining these keystones throughout stands in which they naturally occur, rather than in isolated reserves.

CRITERION 3. LITTLE-KNOWN TAXA

The much longer list of threatened species in Sweden than in British Columbia reflects not only the longer history of intensive forest use, but also the much greater knowledge of the biota of that country. Many of the endangered species in Sweden would not have been identified if they were in British Columbia. Our knowledge is particularly weak on invertebrates, lichens, bryophytes, and fungi, which form most endangered Swedish species (Berg et al. 1994). Ignorance of a species threatens its existence, and can also produce costly surprises to forest managers when a little-known species suddenly becomes a management issue (e.g., northern spotted owl, marbled murrelet).

Exploratory studies of ground-dwelling arthropods at Sicamous Creek and Opax Mountain demonstrated how much knowledge we lack, and the potential for surprises. One planned component of the study identified beetles to family and the carabid beetles to species; 52 families and over 50 species within the family Carabidae indicate the huge diversity within this single order. Even among the relatively well-known carabids, the response of many species to forest harvesting was unstudied, and several better known species did not respond to harvest treatments in the way expected from the few previous studies (Lavallee in prep.).

Pitfall trapping at Sicamous Creek also unexpectedly revealed many grylloblattids (*Grylloblatta campodeiformis*), members of a rare order of winter-specialist insects for which Canada has a high responsibility. No quantitative information on forest management effects was available for any species in the order before our work. The species showed seasonal declines in large clearcuts, but a preference for edge habitats and edge-rich patch-cut arrays (D. Huggard and W. Klenner unpublished data).

Jumping slugs (*Hemphillia* spp.), candidates for listing in the United States because of an association with old forest, are poorly known from British Columbia. However, they were also unexpectedly found to be widely distributed across all treatments at Sicamous Creek. At Opax Mountain, they were restricted to canopied wetter sites, mirroring the U.S. concern for these species in managed drier forests.

We expect that if funding were available in British Columbia for work on poorly known taxa, many more examples would be found of groups that are sensitive to some forest practices, but that could easily be retained in managed forests—if the knowledge of critical habitats were available. At Sicamous Creek and Opax Mountain, the replicated experimental designs and intensive sampling schemes allowed us to make useful conclusions about little-known taxa, even if these taxa were rare or unexpected in the studies.

CONCLUSION

Projects like those at the Sicamous Creek and Opax Mountain silvicultural systems sites help provide applied information that will keep species and habitat elements off our "at risk" lists. Investment in obtaining such information now should pay off by preventing the need for expensive, contentious, and risky preservation or recovery programs in the future. An integrated, prioritized, and proactive provincial research program on species sensitive to forest management is required to avoid the long lists of truly threatened species found in areas with longer histories of management. As priorities, we suggest species or habitat elements threatened or lost in those areas, ecological "keystones," and little-known taxa. We also suggest that priority be given to groups for which the province has a high global responsibility. Finally, we emphasize the importance of accepted study and sampling designs for scientifically valid, useful information, even on rare or unexpected species.

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Development of a Northern Goshawk Habitat Suitability Index for Forest Types of the Kootenay Region

Marlene M. Machmer
Pandion Ecological Research Limited
705 Stanley Street, Nelson, BC, V1L 1N6 Canada
mmachmer@netidea.com

Greg F. Utzig
Kutenai Nature Investigations Limited
602 Richards Street, Nelson BC, V1L 1N6 Canada

Tom M. Gaines
Kutenai Nature Investigations Limited
602 Richards Street
Nelson BC, V1L 1N6 Canada

Carl Savignac
Pandion Ecological Research Limited
705 Stanley Street
Nelson, BC, V1L 1N6 Canada

Jakob Dulisse
Pandion Ecological Research Limited
705 Stanley Street
Nelson, BC, V1L 1N6 Canada

ABSTRACT

The northern goshawk (*Accipiter gentilis*) is a rare to uncommon species in British Columbia and throughout the holarctic. The interior subspecies (*A. gentilis atricapillus*) has been declared "Identified Wildlife" and therefore requires special management attention, including the establishment of wildlife habitat areas (WHAs). We recently developed a habitat suitability index (HSI) to investigate the effects of current forest management practices on northern goshawk habitat in southeastern British Columbia. The index describes goshawk habitat requirements based on a spatially hierarchical framework of nested components (i.e., nest area, post-fledging family area, foraging area) within a home range. Without site-specific information, we used goshawk habitat indicator and criteria values in the HSI from the literature and adjusted them to reflect ecosystem-dependent natural disturbance regimes, forest types, and stand dynamics. This study summarizes empirical data gathered on goshawk breeding habitat use in southeastern British Columbia to further develop the nest area component of the HSI. We conducted call playback surveys and intensive nest searches in the Enhanced Forest Management Pilot Project (EFMPP) area of the Invermere Forest District during July 1998. In areas with confirmed breeding activity, we described the characteristics of active and alternate nest trees and assessed the breeding habitat using variable-radius prism plots in 8 randomly selected sites within each nest area. Many historical northern goshawk nest areas in 4 biogeoclimatic zones of the Kootenay Region were similarly assessed. We summarize empirical data on goshawk breeding habitat characteristics and use this information to recommend ecosystem-specific improvements to the nest area component of the HSI. We also discuss possible practical applications of HSI model output to the Invermere EFMPP area.

Key words: *Accipiter gentilis*, breeding habitat, habitat suitability index, Kootenay, nest trees, northern goshawk.

The northern goshawk (*Accipiter gentilis*) is receiving increased attention from resource managers because of an association with mature and old-growth forests throughout its range. This species has recently been declared "Identified Wildlife" under the British Columbia Forest Practices Code and therefore requires special management attention, including the establishment of wildlife habitat areas (WHAs).

To evaluate the effects of forest management practices on the availability of goshawk habitat in southeastern British Columbia, we recently developed a northern goshawk habitat suitability index (HSI). The index is based on a framework of nested components (i.e., nest area, post-fledging area, foraging area; Fig. 1) within an overall home range. Without information for the interior subspecies (*A. gentilis atricapillus*) in southeastern British Columbia, we used goshawk habitat indicator and criteria values from the literature and adjusted them to reflect local forest types, stand dynamics, and natural disturbance regimes (Table 1).

We summarize preliminary empirical data gathered at active and historical goshawk nest areas in 4 biogeoclimatic variants of the Kootenay Region. These data will be used to further develop the nest area component of the HSI.

METHODS

Call Playback Surveys and Nest Searches

We conducted call playback surveys and intensive nest searches from 13 to 20 July 1998 in the Enhanced Forest Management Pilot Project (EFMPP) area of the Invermere

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1. Proposed habitat indicators and criteria for northern goshawk nest areas in southeastern British Columbia.

Habitat indicator	Weighting	Criteria and level			
		Good (3)[a]	Moderate (2)	Poor (1)	Unsuitable (0)
Very large trees 60 cm	1.0	>50/ha	30–50/ha	10–30/ha	<10/ha
Large trees 40 cm	1.0	>60/ha	40–60/ha	20–40/ha	<20/ha
Crown closure	1.0	>75%	60–75%	50–60%	<50%
Canopy complexity	1.0	>2 layers	2 layers	1 layer	no canopy
Slope	0.5	<30%	30–45%	45–60%	>60%
Aspect	0.3	SW, S, SE, nil	W, E	NW, N, NE	—
Patch size	0.8	>40 ha	20–40 ha	5–20 ha	<5 ha
Distance to water source	0.5	<0.5 km	>0.5–2 km	>2–4 km	>4 km
Distance to active road	0.3	>1,000 m	500–1,000 m	250–500 m	<250 m

[a] Numbers in parentheses are suitability values associated with each level of criteria; nest area suitability (a weighted arithmetic mean of individual indicator values) values range from 0 to 1 and correspond to the following subjectively defined classes: good, >0.8; moderate, 0.6–0.8; poor, 0.4–0.6; unsuitable, <0.4.

Forest District (Fig. 2). Areas with high goshawk nesting potential within the study area were prioritized for surveys using a themed geographic information system (GIS) map generated by British Columbia Ministry of Forests staff, based on the following criteria: >40% crown closure, age class 6–9, <50% slope, and south or west aspects.

Assessment of Active and Historical Nest Areas

We assessed the following characteristics at 18 active and alternate nest trees and sites: 1) nest tree species, diameter at breast height (dbh) (cm); height (m); decay class, defects and disturbance agents; nest height (m), and position; and 2) age class, forest cover type, slope position, elevation (m), site series, and location (with global positioning system [GPS]). In plots centred on active and alternate nest trees and in 8 other randomly chosen locations ≤ 200 m distance from nest sites), we sampled the following parameters: 1) species, dbh (cm), height (m) and decay class of all trees >7.5 cm dbh in variable-radius prism plots; 2) percent crown closure and number of tree layers in 11.28-m fixed-radius plots; and 3) percent slope and aspect. Patch sizes and distances to permanent water and active roads were estimated from air photos and field reconnaissance. We also assessed goshawk breeding habitat at 7 historical nest areas in the Interior Douglas-fir (IDF), Montane Spruce (MS), and Engelmann Spruce–Subalpine Fir (ESSF) biogeoclimatic zones of the East Kootenay. Data gathered at 4 historical nest areas in the Interior Cedar–Hemlock (ICH) zone of the West Kootenay (as part of a separate study) are included for comparison.

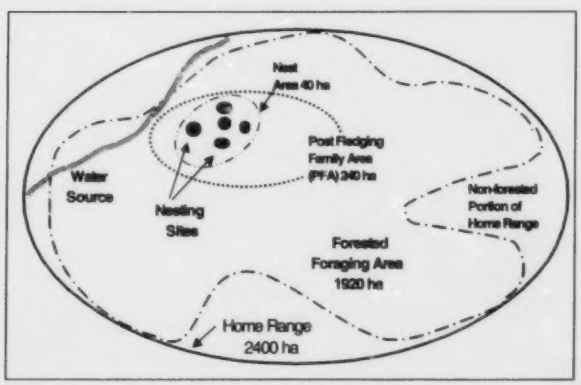


Figure 1. Schematic diagram of northern goshawk habitat requirements.

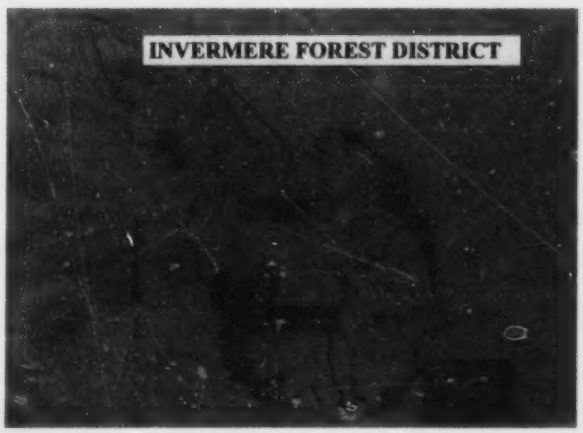


Figure 2. Location of the Enhanced Forest Management Pilot Program (EFMPP) area, Invermere Forest District.

Table 2. Characteristics of active and alternate nest trees and sites found in the East and West Kootenay (mean ± SE [range], unless otherwise indicated).

	East Kootenay (n = 18)	West Kootenay (n = 4)
Nest trees		
Species (%)	56%Fd; 44%Lw	40%Fd; 20%Pl; 20%Pw; 20%Py
Dbh (cm)	54.8 ± 0.2 (36.3–80.0)	63.1 ± 2.2 (38.9–81.0)
Height (m)	29.4 ± 2.1 (26.0–34.0)	33.9 ± 3.3 (27.0–46.0)
Decay class (%)	56% 2; 44% 1	25% 1; 50% 2; 25% 3
Nest height (m)	14.0 ± 0.8 (7.5–22.5)	16.9 ± 2.7 (9.0–24.0)
Nest sites:		
Elevation (m)	849–1,373	799–972
Slope position (%)	44% mid; 33% low; 17% level; 6% upper	67% mid; 33% low
Age class (%)	40% 8; 20% 6 & 7; 10% 5 & 4	50% 8; 33% 6; 17% 7
Forest cover	Fd, Lw, Pl, Sxw, and Bl	Fd, Pl, and Lw

RESULTS

Call Playback Surveys and Nest Searches

Three goshawk responses were received at 198 call stations (response rate = 1.52/100 stations) and fledgling begging calls were heard at 2 additional stations before playing taped calls. These 5 detections resulted in the discovery of 3 active goshawk nest areas in the pilot area. New active nests were also found in 2 historical nest areas. Productivity was 1.8 ± 0.2 (mean fledged birds ± standard error [SE]) at 5 successful nests in the East Kootenay.

Assessment of Active and Historical Nest Areas

Characteristics of active and alternate goshawk nest trees and sites are summarized in Table 2. All goshawk nests were found close to the bole in the lower third of tree crowns. Nest trees were among the largest (upper 10% of diameter and height distribution) available in the stands sampled and 8 of 10 nest trees found in western larch had symptoms of dwarf mistletoe. With the exception of 2 nest sites in the Soowa Mountain nest area (in an age class 4 fire origin lodgepole pine stand bordering a >100-ha, age class 8 patch), nest sites were found at middle, lower, or level slope positions in age class 5–8 stands.

Results of assessments for 9 habitat indicators at 14 northern goshawk nest areas are summarized in Table 3. Nest areas were found on a range of aspects (south and west aspects accounted for >70%) with relatively gentle slopes (≤ 40%), single- to multilayered canopies, 44–76% crown closure, sparse understories, and variable densities of large trees.

In general, nest areas sampled in the West Kootenay had higher suitability values based on the proposed HSI.

We recommend the following site-specific improvements to the HSI until additional data can be obtained: 1) reduce the very large tree diameter limit from >60 to >50 cm; 2) adjust criteria for the very large tree habitat indicator (>20, 5–20, 3–5, and <3 stems per hectare for good, moderate, poor, and unsuitable ratings, respectively) in forest types of the East Kootenay; 3) modify criteria for the percent crown closure habitat indicator (>55 good, 45–55 moderate, 40–45 poor, and <40% unsuitable for East Kootenay; >60 good, 50–60 moderate, 40–50 poor, and <40% unsuitable for West Kootenay), and 4) remove the distance to active road habitat indicator because intensity of road use varies seasonally and over time making it difficult to estimate this parameter.

More intensive northern goshawk field surveys and breeding habitat assessments are planned in the Invermere Forest District for the 1999 breeding season.

APPLICATION OF THE HSI

We recently used the Simulation Forest (SIMFOR) model to evaluate effects of changes in coarse spatial patterns of forest cover on the quantity and quality of northern goshawk foraging habitat in the Invermere EFMPP area of southeastern British Columbia. SIMFOR uses harvest schedules from an external source to define future stand ages for a set of user-mapped stand types. Habitat quality and species response in the model are interpreted by employing user-defined relationships between: 1) the supply of habitat attributes (e.g., crown closure and density of large snags, trees, and coarse woody debris were used for goshawk foraging areas) in different stand ages and types ("habitat supply curves"; Fig. 3); and 2) the relative importance of those attributes to a given species ("habitat suitability curves"; Fig. 4). The degree to which the predicted levels of attributes in a given stand at a specified time match a user-defined optimal habitat determines relative habitat suitability for that site. SIMFOR provides maps and data summaries of habitat quality over time for a site-specific stand or home range (1,920 and 1,440 ha forested foraging areas were used in the simulation runs).

Output maps (examples in Fig. 5) indicated a short to medium term decrease in goshawk habitat suitability with

Table 3. Results for 9 habitat indicators at 14 northern goshawk nest areas in the East and West Kootenay.

Nesting area	Habitat indicators								
	Very large trees (stems/ha)	Large trees (stems/ha)	Crown closure (%)	Canopy layers (no.)	Slope (%)	Aspect	Patch size (ha)	Water source (m)	Active road (m)
EAST KOOTENAY									
IDFdm2									
Whiteswan Lake	11	124	49	2-3	0	nil	61	800	100
Nine Mile Creek	1	45	52	2	19	W	26	750	125
Parson	37	102	57	1	31	[illegible]	109	450	45
Newgate	1	29	62	1	22	[illegible]	164	1,200	750
Soowa Mountain	1	3	69	2	39	N	123	450	[illegible]
Horse Creek	5	48	[illegible]	2	20	SW	57	500	200
MSdk1				2					
Whitetail Lake	1	[illegible]	44	2	29	W	[illegible]	2,000	50
Cedrus Creek	4	119	[illegible]	2	13	W	268	400	[illegible]
Maryann Creek	[illegible]	78	[illegible]	2-3	25	SW	[illegible]	250	1,500
ESSFdk									
Munroe Lake	2	50	46	2	23	SW	116	400	[illegible]
WEST KOOTENAY									
IDFdw									
Koch Face	20	313	60	2	40	SE	67	100	100
ICHmw2									
Rusty Creek	10	83	43	2	40	SE	118	250	200
Whatshan Lake	55	300	76	2-3	20	W	[illegible]	250	1,500
Merry Creek	20	155	66	2	23	E	180	700	[illegible]
Suitability	Unsuitable		[illegible]			Moderate		Good	

subsequent increases 50–100 years from now. Differences between site and home range level habitat suitability indicated that although numerous individual stands have good quality habitat, many are too fragmented to provide a sufficient area of high quality habitat within individual home ranges. This result may be partly caused by the use of circular home ranges in the model (elongated home ranges may have provided more realistic habitat suitability estimates for the narrow valleys and intervening alpine zones found in the study area). Sensitivity analysis showed that the subjective setting of class limits for habitat quality may be one of the most important factors in the overall habitat assessment outcome.

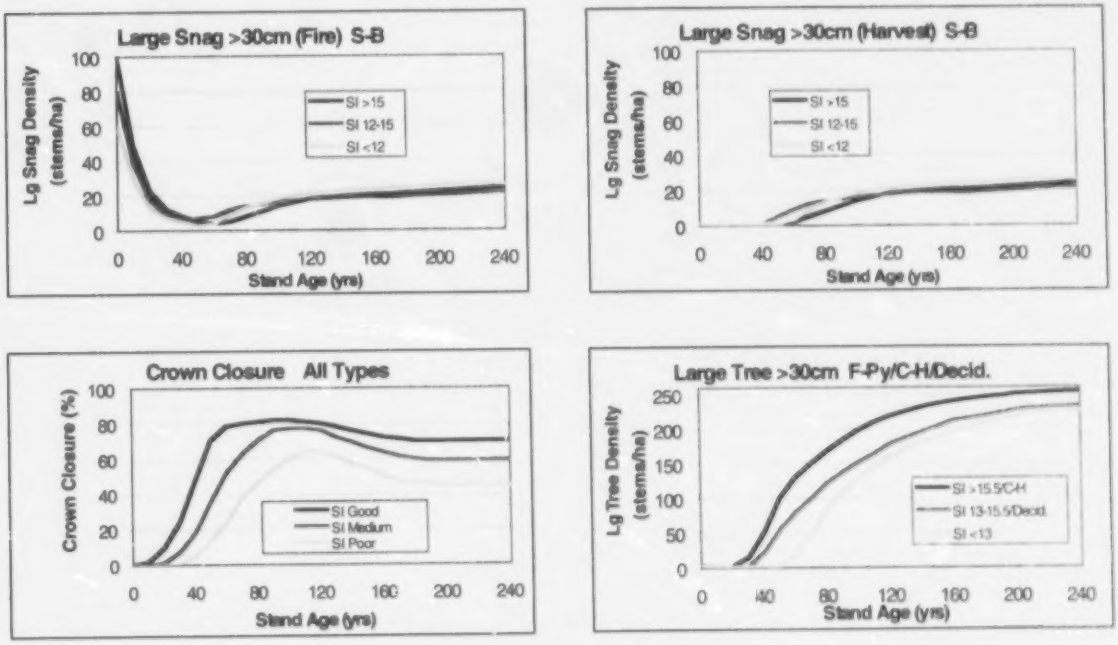


Figure 3. Examples of northern goshawk foraging habitat supply curves at 3 suitability index classes.

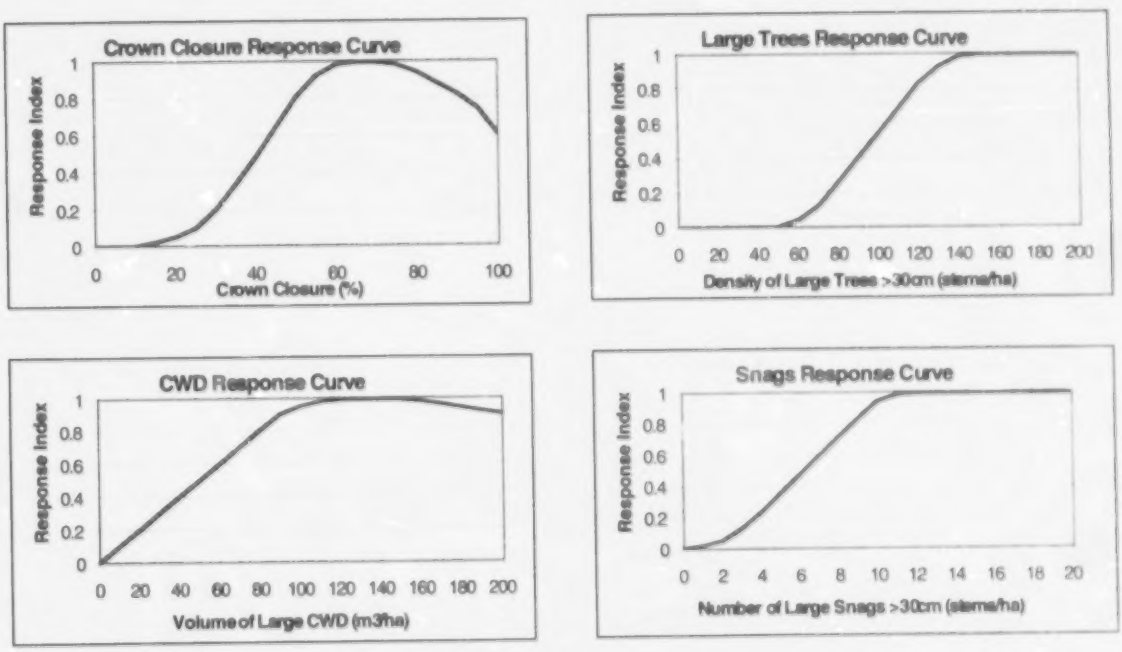
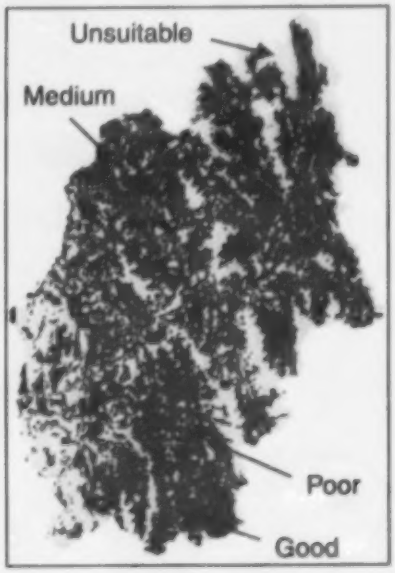


Figure 4. Examples of northern goshawk foraging habitat suitability curves.



Site level foraging habitat suitability – Year 100.

Home range level foraging habitat suitability – Year 100 (HR = 1920 ha, class limits 0.3, 0.5, 0.7).

Home range level foraging habitat suitability – Year 100 (HR = 1920 ha, class limits 0.4, 0.6, 0.8).

Home range level foraging habitat suitability – Year 100 (HR = 1440 ha, class limits 0.4, 0.6, 0.8).

Figure 5. Examples of northern goshawk foraging habitat suitability output for the EFMPP area generated by the SIMFOR habitat analysis module.

Status and Conservation Stewardship of the Pacific Great Blue Heron in Canada

Robert W. Butler
Environment Canada, Canadian Wildlife Service
5421 Robertson Road, Delta, BC, V4K 3N2, Canada
rob.butler@ec.gc.ca

Philip D. Baudin
WBT Wild Bird Trust of British Columbia
124 - 1489 Marine Drive, West Vancouver, BC, V7T 1B8, Canada

ABSTRACT

The distribution of the Pacific subspecies of great blue heron, *Ardea herodias fannini*, is confined to the Pacific coast of North America from Prince William Sound, Alaska, south to Puget Sound, Washington, where it resides year-round. About 4,000–5,000 pairs are estimated to occupy this range, of which about 2,000 reside in British Columbia. There are about 20–30 colonies of herons around the Strait of Georgia and about 80% of the herons nest in 4 colonies near urban areas. The Strait of Georgia population has declined about 3–5% per year since the 1960s. The decline is thought to be a consequence of delayed and failed breeding arising from disturbance by humans and bald eagles (*Haliaeetus leucocephalus*).

Key words: *Ardea herodias fannini*, great blue heron, population trend, status, Strait of Georgia.

The great blue heron (*Ardea herodias*) breeds in south-coastal and southeastern Alaska, coastal and southern British Columbia, northern Alberta, central Manitoba, southern Ontario and Quebec, New Brunswick, Prince Edward Island, and Nova Scotia, south throughout the United States and coastal Mexico, and on the Galapagos (American Ornithologists' Union 1983, Butler 1992). The winter distribution is from south of the frozen regions in the north to as far south as Panama. Five subspecies are recognized within this range (Payne 1979). The *fannini* subspecies, the subject of this paper, is confined to the Pacific coast from Prince William Sound, Alaska, south to Puget Sound, Washington. The Canadian distribution encompasses the entire coast of British Columbia, including Vancouver Island and the Queen Charlotte Islands. The *herodias* subspecies breeds in the interior of British Columbia and across Canada. The Pacific subspecies, *fannini*, is darker in plumage and its populations are more sedentary compared to the pale-plumaged and largely migratory *herodias*.

The great blue heron has high public appeal as a symbol of wetland conservation and environmental quality. This species is protected from hunting and wanton disturbance by the Migratory Birds Convention Act, Migratory Birds Regulations, and the British Columbia Wildlife Act. The species inhabiting British Columbia is on the provincial Blue List of vulnerable species compiled by the British Columbia Ministry of Environment, Lands and Parks, and the Pacific subspecies is listed as Vulnerable by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC).

Three lines of evidence suggest that heron populations are threatened on the south coast of British Columbia. First, Breeding Bird Surveys (BBS) indicate significant declines of about 3–5% per year in populations of this subspecies between 1966 and 1994. Second, habitat occupied 25 years ago has remained vacant (Butler 1997). Third, nest occupancy in disturbed heron colonies was significantly lower than in colonies with little or no disturbance (Werschkul et al. 1976).

PROTECTION

The nests, eggs, young, and adults are protected from hunting by Article 11:3 of the Migratory Birds Convention Act (1994) and Section 5(4) of the Migratory Birds Regulations. Section 6(a) of the Migratory Birds Regulations forbids the disturbance, destruction, and taking of nests or eggs. The BC Wildlife Act (1982) forbids the alteration or destruction of wildlife habitat within Wildlife Management Areas (Section 7[1][a]), hunting endangered or threatened species (Section 26[1][a]), hunting outside of open season (Section 26[1][c]), harassment with motor vehicles, boats, or aircraft (Section 27[3]), and possession or injury of birds or their nests (Section 34). Scare/kill permits were provided up to 1998 where herons fed on fish stocks, but they have since been revoked.

POPULATION SIZE AND TREND

The worldwide breeding population of *A. h. fannini* is estimated to be 4,000–5,000 pairs, of which about 2,000 breed in southwestern British Columbia (Butler 1997). Censuses in

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Puget Sound have not been as thorough as in British Columbia, but Butler (1997) estimated the Washington population to be about 2,000 pairs. The coast outside the Strait of Georgia and Puget Sound has not been formally censused for herons because no large concentrations are known to occur there during the breeding season. One colony of 9 pairs found near Tahsis in 1989 is the only colony reported from the west coast of Vancouver Island. The Tofino mudflats have supported up to about 100 herons in August each year but few stayed during the breeding season (A. Dorst, Tofino, pers. comm.). Butler (1997) estimated that about 400 pairs reside elsewhere along the coast of British Columbia and Alaska.

BBS data indicate a significant ($p < 0.05$) annual decline in herons on the British Columbia coast of 5.7% over the period 1966–94 (Downes and Collins 1996). However, the habit of herons to relocate colonies following a disturbance could explain a sudden decline in detections along BBS routes. Nevertheless, some regions have fewer herons now than in the recent past. In the 1970s and 1980s, the Sunshine Coast had 2 colonies, with 44 pairs near Sechelt and 94 pairs in Pender Harbour (Butler 1997). These colonies were abandoned after disturbances and relocated colonies have not been found despite repeated searches. About half of all colonies around the Strait of Georgia abandoned sites between 1987 and 1992 or suffered a significantly reduced reproductive success (Butler et al. 1995). Three large colonies with more than 100 nesting pairs are located near Chilliwack (approx 125 pairs), in Pacific Spirit Park in Vancouver (approx 140 pairs), and on Salt Spring Island (approx 122 pairs). A fourth colony (approx 350 pairs) nests about 1 km south of the Canada–United States border on Point Roberts, Washington, and forages entirely in British Columbia. A colony near Nanaimo that held about 120 pairs for many years declined to about 80 pairs by 1998. Butler (1997) showed that the source of new colonies could be explained by natal recruitment of young. About 36% of the Canadian population (approx 733 pairs) now nests in 4 large colonies (listed above), which increases their vulnerability to a single disturbance event.

HABITAT AND BIOLOGY

The habitat of the great blue heron was described by Butler (1995, 1997). The breeding habitat includes a wide variety of tree species, where nests are built often in colonies and within 8 km (most within 3 km) of coastal foraging habitats. The largest number of herons nest near large eelgrass (*Zostera marina*) meadows, along rivers, and in estuarine and freshwater marshes, where they feed on small fish. Smaller numbers of herons forage in kelp forests, from wharves, and on shallow beaches. In autumn, juvenile herons on the Fraser River delta occupy grasslands, where they eat small rodents, and elsewhere reside along coastal shores, where they eat fish. Adults occupy estuarine marshes, riverine marshes, and grasslands on the Fraser River delta and coastal shores elsewhere in the province. Abandoned grasslands with small mammal populations are uncommon on the Fraser River delta, whereas other habitats are widespread on the delta and along the coast. Not all habitat is used by herons each year, indicating that population growth is limited by factors other than foraging habitat.

The quality of heron nesting habitat might be declining as a result of increased disturbance by humans and bald eagles (*Haliaeetus leucocephalus*). Most herons nest in quiet forest areas near foraging habitats. These sites have become increasingly scarce in southern British Columbia as a result of the rapid immigration of humans to the region. The human population around the Strait of Georgia is projected to double in the next 30 years. Under present conditions we can expect increased human disturbance at heron colonies.

The breeding biology of this subspecies is described in detail by Butler (1997). Briefly, herons arrive at colonies in February and March, lay first clutches in March and April, and raise chicks from late April to mid-August. The mean clutch size is about 4 eggs and fledging success per egg is about 47%, comparable to other North American studies (Butler 1997). Some industrial contaminants reduce chick growth and eggshell thickness (Elliott et al. 1989, 1996; Whitehead 1989) but these are local effects. Little is known about the age structure or sex ratio of the population. It is assumed that all adults breed, since large concentrations of nonbreeding herons are not evident.

CONSERVATION ISSUES

Some herons can tolerate human activities near their nests, but many are sensitive to the presence of humans. The combined effects of logging and urbanization around the Strait of Georgia have resulted in fewer places for herons to nest that are not subject to human activity. Butler et al. (1995) and Carlson and McLean (1996) showed that the number of fledglings raised in heron colonies with frequent disturbances was significantly lower than at colonies with no disturbance. Herons leave nests unguarded when humans enter colonies on foot or when loud noises occur nearby, especially early in the nesting season. Corvids are quick to take eggs when the opportunity arises (Moul 1990) and their populations have grown in the Fraser River delta (Butler and Campbell 1987).

Factors that possibly limit the maintenance of heron populations are: 1) breeding failure of adult herons that are regularly disturbed by eagles or by human activity near the colonies (Butler et al. 1995); 2) reduced juvenile survival as a result of food shortages in autumn (Butler 1994); and 3) shooting of herons. Many forested sites where herons can nest remain, but few are undisturbed by humans or eagles (Norman et al. 1989). Disturbance results in significantly fewer chicks being raised (Butler et al. 1995) and might be

the reason not all habitat is occupied. Bald eagles are implicated in some of this disturbance, when they prey on adult herons and their chicks at nests. An increase in the number of eagles on the British Columbia coast has led to increased disturbance of heron colonies (Vermeer et al. 1989, Butler 1995). Little can be done to reduce eagle disturbance, but guidelines are available to reduce human disturbance near heron colonies (reviewed by Butler 1997). Physical barriers are the most effective deterrent to human disturbances of heron colonies (Carlson and McLean 1996) and need to be considered around woodlots that hold nesting colonies. Shooting of a small number of herons that were considered pests at fish hatcheries was permitted until 1998. However, in areas with small populations of herons, hatcheries might have acted as sinks attracting herons, especially juveniles, that were subsequently shot, thereby preventing heron populations from growing. This hypothesis is conjecture and requires further evaluation.

HERON STEWARDSHIP PROGRAM

The Heron Stewardship Program (HSP) is a partnership between Environment Canada, the British Columbia Ministry of Environment, Lands and Parks, and the Wildlife Bird Trust of British Columbia. It was launched in 1997 to secure important heron habitats on the coast of British Columbia. Funds for the project were obtained from the Habitat Conservation Trust Fund and Environment Canada's Georgia Basin Ecosystem Initiative. The HSP, with assistance from the Waterbird Watch Collective, purchased 1 colony site on Salt Spring Island, and is working on agreements with owners to manage heron colonies at Chilliwack, Pacific Spirit Park, Point Roberts, Burnaby, and Sooke.

ACKNOWLEDGEMENTS

G. Dick and C. Copland provided information on the legislation to protect herons. This research was funded by the Habitat Conservation Trust of British Columbia, and the Canadian Wildlife Service. M. Chutter, K. Morrison, and D. Dunbar reviewed the manuscript.

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Northern Goshawk Population Inventory for Vancouver Island, British Columbia, 1994–1998

Erica McClaren[1]

Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
2080A Labieux Road South, Nanaimo, BC, V9Y 6J9, Canada

ABSTRACT

Northern goshawk (Queen Charlotte subspecies: *Accipiter gentilis laingi*) inventory and research occurred on Vancouver Island, B.C. from 1994 to 1998. Broadcast surveys, the most effective inventory method known for goshawks, continue to be time-consuming and costly, with a relatively low number of goshawk detections and nest locations per unit of effort. Since 1994, approximately 2,912 broadcast stations resulted in 51 goshawk detections and the location of 19 nest territories. Twelve other goshawk territories were reported by forestry workers and members of the public, 3 were found using radiotelemetry, and another was discovered using the stand-watch inventory technique, increasing the number of currently known goshawk territories on Vancouver Island to 35. Territory occupancy rates appear to vary by year, and may relate to the seral stage of the nesting habitat and to the degree of fragmentation in the surrounding landscape. Research efforts on Vancouver Island have focused on describing the habitat characteristics of goshawk nest sites, nest stands, and home ranges, and on identifying their main prey items. As well, radiotelemetry is being used to gain information on goshawks' breeding and winter home range size and habitat use, territory and mate fidelity, breeding dispersal, and mortality rates. Future directions for goshawk research on Vancouver Island include: 1) testing the effectiveness of a new broadcast call for goshawk inventory; 2) testing the effectiveness of dawn vocalizations to determine territory occupancy and to locate active nest sites; and 3) examining nest productivity and territory occupancy in relation to the percent of old-growth and degree of fragmentation in goshawk breeding home ranges. This research aims to improve our ability to locate goshawk nest territories, gain a better understanding of population size on Vancouver Island, and increase our knowledge of habitat quality for goshawks. Using this information, current management strategies for goshawks may be refined to reflect the conclusions of local, scientific studies.

Key words: *Accipiter gentilis laingi*, inventory, northern goshawk, population, Vancouver Island.

Northern goshawks (*Accipiter gentilis*) are the largest member of the genus *Accipiter* in North America. Like other accipiters, such as the sharp-shinned hawk (*A. striatus*) and Cooper's hawk (*A. cooperii*), goshawks possess a long, rudder-like tail and relatively short, rounded wings, which provide them with both rapid acceleration and excellent maneuverability in their forested habitats. Regional variation among goshawks in British Columbia has resulted in 2 recognized subspecies: *A. g. atricapillus*, which occurs widely throughout British Columbia's mainland and North America; and *A. g. laingi*, referred to as the Queen Charlotte goshawk, which inhabits a much more localized range along the Pacific coast from Alaska to Washington (Squires and Reynolds 1997). Within British Columbia, Queen Charlotte goshawks, the focus of this study, appear to be isolated to the coastal islands, primarily Vancouver Island and the Queen Charlotte Islands (Campbell et al. 1990).

The conviction that northern goshawks require mature and old-growth forests to meet their life history requirements has placed them at the forefront of forest harvest and land-use conflicts in Europe (Kenward and Widen 1989, Tommeraas 1994, Widen 1997) and the United States (Crocker-Bedford 1990, Clark 1998) for over a decade and more recently in Canada (Duncan and Kirk 1995). However, goshawks are a difficult raptor to detect in their forested habitats due to their secretive behaviour, relatively low population densities, and large home range size (Kennedy and Stahlecker 1993, Joy et al. 1994, Bosakowski and Vaughn 1996). Consequently, there is a paucity of information regarding the impact of forest harvest and fragmentation on goshawk populations. Goshawks appear to select for habitat

[1] Present address: Fishery and Wildlife Biology, Colorado State University, Ft. Collins, Colorado mcclaren@lamar.colostate.edu

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

based on structure rather than on type, typically nesting and foraging in mature and old-growth deciduous, coniferous, and mixed-wood forests characterized by large volume trees and closed canopies (Squires and Reynolds 1997, Daw et al. 1998). Goshawks prey upon a variety of species, occupying the role of a generalized, opportunistic hunter (Squires and Reynolds 1997), although structural characteristics of forests may dictate prey availability (Beier and Drennan 1997, Widen 1997).

Concerns regarding the population status of goshawks have resulted in petitions to list *A. g. atricapillus* populations west of the 100th meridian as threatened under the United States Endangered Species Act (ESA; Clark 1998). After reviewing the published scientific literature, Kennedy (1997) reported that there was no evidence to support the claim that goshawk populations were declining in the western United States. Kennedy (1997) stated that this result could be interpreted in 2 ways: 1) that goshawk populations are not declining; or 2) goshawk populations are declining but current sampling techniques are unable to detect this trend. Likewise, in 1998, the petition to list the goshawk under the ESA was rejected, due to insufficient evidence of population declines (Clark 1998).

More recently, conservation concerns have shifted to the Queen Charlotte goshawk. Petitions to list this subspecies as threatened or endangered under the American ESA, for similar reasons as described above for *A. g. atricapillus*, are still being debated in the courts. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) designates the northern goshawk as Not at Risk and the Queen Charlotte goshawk as Vulnerable (Duncan and Kirk 1995). Within British Columbia, northern goshawks are not listed, whereas Queen Charlotte goshawks occur on the provincial Red List as a candidate for threatened or endangered status (B.C. MELP 1998). Furthermore, goshawks (both subspecies) are an Identified Wildlife Species under the British Columbia Forest Practices Code Act (Forest Practices Code of BC 1996) and Queen Charlotte goshawks are ranked by the provincial Conservation Data Centre as S2B, SZN (imperilled in British Columbia due to rarity and perceived threats to habitat). Due to the paucity of information on Queen Charlotte goshawk habitat requirements, it is unclear how habitat data collected on northern goshawks in the western United States apply to this subspecies (Daw et al. 1998).

Management guidelines have recently been developed in British Columbia for both subspecies of goshawks, in *Managing Identified Wildlife: Procedures and Measures, volume I* (B.C. MOF and B.C. MELP 1999) under the British Columbia Forest Practices Code Act (Forest Practices Code of B.C. 1996). However, these guidelines have been primarily developed from studies in the United States on *A. g. atricapillus*. Consequently, it is unknown how accurately these guidelines reflect the habitat requirements of Queen Charlotte goshawks on Vancouver Island. In order to gather more information on Queen Charlotte goshawks, the Wildlife Branch of the British Columbia Ministry of Environment, Lands and Parks (MELP) initiated goshawk research and inventory on Vancouver Island in 1994 and 1995, respectively. While there is often no clear distinction between inventory and research, these efforts have focused on: 1) distribution and breeding densities; 2) breeding and winter habitat associations, home range size, and movement patterns; 3) prey species composition; 4) territory occupancy and nest productivity in relation to the forest age class and the degree of fragmentation within home ranges; 5) nest site and mate fidelity; and 6) survivorship. Due to the length restriction of this paper, none of the above topics will be discussed in depth and some will not be discussed at all. Please refer to Ethier (1999) for a more detailed discussion of goshawk habitat associations and prey composition on Vancouver Island, and to McClaren (1999) for more information on other topics that are not included in this paper.

STUDY AREAS

Although inventory efforts have concentrated in the Woss/Sayward areas of Vancouver Island, inventory has also occurred in other areas, including Cowichan Lake, the Walbran Valley, Nanaimo Lakes, Port Alberni, Strathcona Park, Campbell River, Gold River, and parts of Quadra Island. Inventory also occurred on the Queen Charlotte Islands between 1995 and 1998, although this is currently organized by the Skeena region of MELP and, therefore, will not be discussed in this report. Inventory efforts have aimed to be unbiased by focusing equal effort in forests representing 3 silvicultural treatments: 1) largely uncut, continuous old-growth forests (>250 yrs); 2) largely uncut, continuous second-growth forests (40–140 yrs); and 3) largely cut, unconnected patches of old-growth and second-growth forests (defined here as fragmented). Since 1997, inventory between Campbell River and Woss has primarily focused on resurveying a 1.6-km radius around goshawk nest clusters within known territories to determine occupancy status and to locate active nest trees.

METHODS

Goshawk Inventory

Standardized inventory methods, outlined by the Resources Inventory Committee (RIC) in Ethier and McClaren (1997) were utilized for goshawk surveys on Vancouver Island. Two types of survey techniques were used in goshawk inventories: broadcast surveys and stand-watches. Kennedy and Stahlecker (1993) originally described broadcast survey protocol for conducting goshawk inventory in forests of the southwestern United States and stand-watches were suggested by Reynolds

(1983). All habitat data collection followed standards outlined in Luttmerding et al. (1990). Refer to McClaren (1999) for a more detailed description of these survey techniques.

Assessing Territory Occupancy and Nest Productivity

Known goshawk nest sites on Vancouver Island were revisited between 1995 and 1998 during the breeding seasons to assess their occupancy status and to gather nest productivity data. All known nests within each territory were assessed for activity using a spotting scope and tripod, to look for an incubating female or for fresh greenery and downy feathers around the nest's edge. Other clues that enabled us to assess occupancy at nest sites included goshawks calling, as well as fresh whitewash and plucking posts in the vicinity of known nest sites. If nests appeared to be inactive, calls were broadcast within a 1.6-km radius around known nest clusters to locate any unknown, alternate nests that may have been active within territories. Goshawk territories that appeared inactive in May and June were resurveyed during late July and early August to reassess the presence or absence of fledglings. If, after a minimum of 2 surveys (1 during the nestling phase, 1 during the fledgling phase), no goshawks were detected, territories were considered unoccupied. If after 2 surveys, 1 or more adult goshawks were detected within a territory but no active nest or fledglings were located, territories were considered occupied but inactive.

Nest productivity was recorded as the number of young observed in the nest approximately 1 week prior to fledging. This definition for nest productivity closely resembles recommendations by Steenhof (1987), who states 39 days after hatching is the appropriate age to determine the number of young successfully fledged from a goshawk nest. It is better to estimate the number of young that reach a certain size in the nest rather than count the number after fledging, since birds are easily missed after they fledge, resulting in lower estimates for nest productivity (Steenhof 1987). Additionally, nest failures were recorded when they occurred. This information will be used to examine differences in territory occupancy and nest productivity relative to forest age class and the degree of fragmentation within goshawk home ranges.

Trapping

Two methods of trapping were used to capture goshawks. One method, referred to as the dho-gaza trap, used a live, permanently injured, non-releasable great horned owl (*Bubo virginianus*) from a wildlife recovery centre, and a mist-net. The great horned owl was tethered to a perch below the mist-net near active goshawk nests. Goshawks' aggressive nature near their nests results in them swooping to scare the owl and getting caught in the net, without causing injuries to themselves or to the owl. Box traps, modified from the original design by Kenward and Marcstrom (1983), were the second method used to capture goshawks. Box traps, with rock doves (*Columba livia*) in the center compartment, lure goshawks into separate side compartments when they are trying to capture rock doves for food. Box traps were placed along road-sides or at clearcut edges next to forested stands where goshawk nests were known or were suspected to be located. Dho-gazas were used in June and early July to capture adults when they were most aggressive around their nests, whereas box traps were used in late July and August to capture young of the year before they dispersed.

Trapped goshawks were hooded and a series of morphometrics and health indicators were recorded (McClaren 1999). Wing and tail moults were also recorded, the birds were weighed and banded, and some were affixed with BIOTRACK® backpack radio-transmitters (BIOTRACK Ltd., UK). Radio-transmitters were affixed to adults; immatures were banded only. When other raptors were caught in box traps, morphometric and health data were recorded, and then they were released.

Radiotelemetry

Aerial telemetry, using helicopters and fixed-wing aircraft, was used to locate goshawks periodically throughout the breeding season and winter months, whenever finances and weather permitted. In May, an effort was made to relocate all tagged individuals (especially females) to determine their nesting territories and nest trees. Once general nest locations were determined by air, ground telemetry was performed on foot using a 6-element antenna (Lindsay, ON) to locate and/or verify active nest trees. On average, air telemetry occurred twice a month, from September through April, 1996–1999. Goshawk locations (UTM [universal transverse mercator] coordinates) were recorded using a GPS (global positioning system) from the aircraft. Additionally, broad habitat data, such as tree species composition, age class and forest stand type, were recorded from the air. UTM coordinates were entered into a spreadsheet in Microsoft Excel® where the distance between an individual's current location and its nest site and last telemetry location were calculated to examine movement patterns. UTM coordinates were also entered into a GIS (geographic information system) database so that telemetry locations could be viewed on maps in relation to landscape-level habitat features. Aerial locations will be used to determine goshawk winter home ranges, winter habitat associations, and survival when sufficient data are available.

RESULTS

Goshawk Inventory

Between May and August, 1994–1998, approximately 2,912 broadcast stations (37,856 ha) resulted in 51 confirmed goshawk detections and the location of 19 goshawk territories (Table 1). Approximately 45% of these call stations occurred

Table 1. Survey effort, goshawk detections, and nest territories found on Vancouver Island in 3 forest types, May–August, 1994–1998.

Survey areas	Forest type[a]	Call stations			Stand-watches		No. of new nests
		No.	Area (ha)	Detections	Hours	Detections	
Vancouver Island	Continuous OG	1,285	16,705	30	494	14	12
	Continuous SG	962	12,506	6	76	1	3
	Fragmented OG/SG	665	8,645	15	162	5	5
OVERALL TOTAL	All Forest Types	2,912	37,856	51	732	20	20

[a] OG = old growth, SG = second growth.

in continuous old-growth forests, 32% in continuous second-growth forests, and 23% in fragmented forests. Of the 51 goshawk detections, 59% occurred in continuous old growth, 12% occurred in continuous second growth, and 29% occurred in fragmented forests.

Stand-watches appear to be a less effective inventory technique, with approximately 732 stand-watch hours conducted between 1995 and 1998 generating 20 confirmed goshawk detections and 1 new nest territory (Table 1).

Additionally, 12 goshawk territories were reported by forestry workers and members of the public, and 3 were found using radiotelemetry, increasing the total number of goshawk nest territories currently known on Vancouver Island to 35.

Territory Occupancy and Nest Productivity

Between 1995 and 1998, occupancy of known goshawk territories on Vancouver Island was consistently higher at nests located in continuous old-growth forests, although a greater number of nests are known within this forest type (Fig. 1). Furthermore, overall occupancy rates varied annually.

As well as assessing known goshawk territories for occupancy, the number of young in each nest approximately 1 week prior to fledging (approx. 39 days of age) was documented (Table 2). Mean productivity between 1994 and 1998 varied by year and by forest type.

Nesting Habitat

Of the 35 goshawk territories located on Vancouver Island between 1994 and 1998, 18 (51%) are in continuous old-growth forests, 11 (32%) are in a continuous second-growth forests, and 6 (17%) are in fragmented forests (Table 3). These territories are distributed fairly evenly among the 4 cardinal directions for stand aspects (Fig. 2). Thus, goshawk nests are situated in forested stands with various slope aspects. The elevational range of known goshawk nest trees is between 80 m for the West Cracroft Island nest and 810 m for the Oktawanch nest, with a mean nest elevation of 375.1±31.4 m for the 35 territories.

At a smaller scale, goshawk nests were in a wide variety of tree species (Table 3). Twenty-nine (53.7%) were in Douglas-fir (*Pseudotsuga menziesii*), 19 (35.2%) in western hemlock (*Tsuga heterophylla*), 6 (11.1%) in red alder (*Alnus rubra*), 1 in a Sitka spruce (*Picea sitchensis*), and 1 in a western redcedar (*Thuja plicata*). Tree species data are missing for 5 nest trees. In general, nest trees had a large diameter at breast height (DBH; mean = 68.6±5.8 cm) and nests were located one-third to one-half the way up the nest

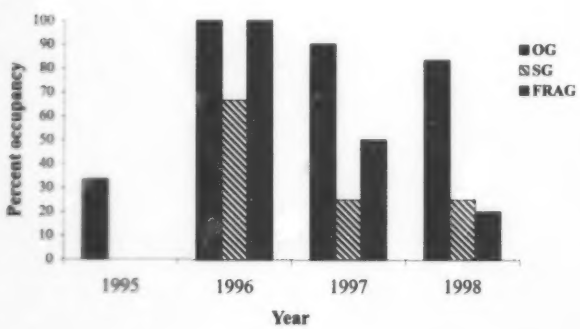

Figure 1. Percent occupancy of known goshawk territories on Vancouver Island, 1994–1998.

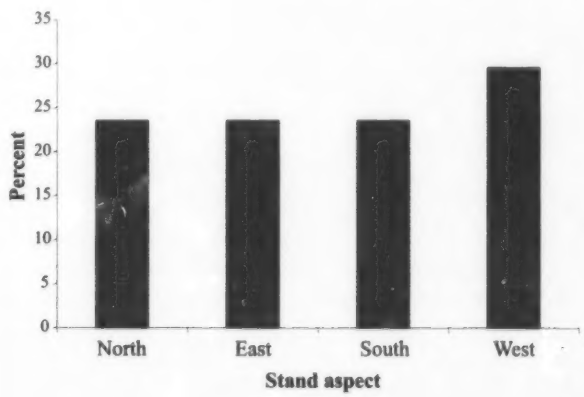

Figure 2. Percent of goshawk nest territories on each stand aspect.

Table 2. Mean nest productivity (number fledglings/active nest), 1994–1998, by year and forest type. Means are followed by standard errors and sample size.

Forest Type[a]	1994 $\bar{\chi}$ ±SE (n)	1995 $\bar{\chi}$ ±SE (n)	1996 $\bar{\chi}$ ±SE (n)	1997 $\bar{\chi}$ ±SE (n)	1998 $\bar{\chi}$ ±SE (n)	Total
COG	2.0±0.4 (4)	2.0±0.0 (4)	2.0±0.2 (10)	1.2±0.4 (9)	1.9±0.2 (12)	1.7±0.1 (35)
CSG	3.0±0.0 (1)	2.0±0.0 (2)	2.0±1.0 (3)	1.7±0.2 (6)	2.0±0.6 (3)	1.9±0.2 (14)
FRAG			0.7±0.5 (4)	2.0±0.0 (2)	2.0±0.0 (1)	1.3±0.4 (7)
Total	2.2±0.4 (5)	2.0±0.0 (6)	1.7±0.2 (17)	1.4±0.2 (17)	1.9±0.2 (16)	1.7±0.1 (56)

[a] COG = continuous old-growth forests; CSG = continuous second-growth forests; FRAG = fragmented old-growth and second-growth forests.

tree. With the exception of 4 nests, most goshawk nests were built in live trees.

DISCUSSION

Goshawk Inventory

Between 1994 and 1998, goshawk inventory occurred in several different areas on Vancouver Island. Inventory work in new areas improves our understanding of the nesting distribution and habitat associations of goshawks on Vancouver Island. Surveys conducted repeatedly within known territories are used to determine territory occupancy status and to locate alternate nest sites. Repeated inventory within the Woss/Sayward research areas enables us monitor goshawk populations in this area over time, to evaluate territory occupancy and nest productivity in relation to forest age class distribution and the degree of fragmentation within breeding home ranges, and to gain more information on nest site and mate fidelity for tagged individuals.

The effectiveness of broadcast surveys for goshawks within the dense, rugged, rain-soaked Pacific Northwest forests has been questioned by several researchers and inventory personnel (western Washington: Bosakowski and Vaughn 1996; southeast Alaska: Iverson et al. 1996; western Oregon: DeStefano and McCloskey 1997). However, in a recent test of the broadcast survey methodology in western Washington, Watson et al. (1999) concluded that the effectiveness of broadcast surveys within Pacific Northwest forests was comparable to the dry, southwestern forests where the methodology was originally described (Kennedy and Stahlecker 1993). Between June and August, 1994–1998, 51 goshawk detections occurred through broadcast surveys on Vancouver Island. Although this appears to be low relative to the number of broadcast stations, it compares well with other studies. For example, a study conducted on a population of goshawks with 1 of the highest known nesting densities in North America, the Kaibab Plateau in Arizona, reported 2.0 detections (nestling phase) and 1.0 detections (fledgling phase)/100 broadcast stations (Joy et al. 1994). Additionally, broadcast survey success has improved annually on Vancouver Island with 0.4, 1.1, 1.6, and 5.2 detections per 100 call stations in 1995, 1996, 1997, and 1998 respectively (Quayle et al. 1995, McClaren 1997). The greater detection rates over recent years probably reflect a combination of increased inventory efforts within known goshawk territories to determine occupancy status and returning employees who have higher skill levels. It is unlikely that increased detection rates reflect greater goshawk numbers over this 4-year period, although it may reflect fluctuations in the percentage of goshawks nesting from year to year. Therefore, broadcast surveys, although time-consuming, labour-intensive, and costly, appear to be the most effective survey technique currently known to detect breeding goshawks. Furthermore, this survey methodology appears to be nearly as effective in the forests on Vancouver Island as it does in the dry forests of the southwestern United States.

Stand-watches, originally described by Reynolds (1983) as an alternate method for conducting goshawk surveys, remain relatively effective for detecting goshawks, but less useful for locating goshawk nest sites. Detection rates were 5.0 (1995), 10.0 (1996), 11.0 (1997), and 6.2 (1998) per 100 stand-watch hours (Quayle et al. 1995; McClaren 1997, 1999). It is difficult to compare detection rates between stand-watches and broadcast surveys since the latter is measured as detections per hour and the former as detections per 100 stations. Regardless, more goshawks are detected per unit of effort with stand-watches than broadcast surveys. However, stand-watches are relatively ineffective for locating goshawk nest sites. For example, stand-watch detections on Vancouver Island led to no new nest sites in 1995, 1997, or 1998, and to only 1 in 1996 (Quayle et al. 1995, McClaren 1997). In comparison, broadcast surveys resulted in 2, 4, 3, and 5 new nest territories from 1995 to 1998, respectively (Quayle et al. 1995, McClaren 1997). Broadcast surveys are more effective for locating nests than stand-watches because, typically, detections are highest within 300 m of an active nest (Kennedy and Stahlecker 1993, Watson et al. 1999). In comparison, goshawk detections during stand-watches may be foraging birds far from their nests, or birds that are not associated with active territories. Additionally, the stand-watch survey technique is more weather-dependent

Table 3. Habitat characteristics of known goshawk nest trees found during the summers of 1994–1998 in continuous old-growth (COG), continuous second-growth (CSG) and fragmented (FRAG) forests on Vancouver Island. (No data [ND] were collected for some parameters due to time constraints and/or equipment limitations).

Nest	Forest type	Tree species[a]	Elevation(m)	DBH[b] (cm)	Nest ht.(m)	Tree ht.(m)	Tree age (yr)	Decay
Artlish	COG	WH	215	47.5	19.0	49.3	>250	Alive
McLaughlin R. #1	COG	WH	700	45.0	20.8	28.6	>250	Alive
McLaughlin R. #2	COG	DF	650	65.0	21.2	36.2	>250	Alive
McLaughlin R. #3	COG	DF	800	75.5	19.0	20.4	>250	Alive
Lower Stella #1	CSG	DF	155	40.2	18.6	34.0	60–80	Alive
Lower Stella #2	CSG	DF	170	51.3	11.9	33.0	60–80	Alive
Loon Lake #1	FRAG	DF	375	41.7	16.3	17.1	>250	Alive
Loon Lake #2	FRAG	DF	300	38.2	ND	ND	>250	Dead
Tlatlos #1	COG	WH	465	66.3	14.7	44.9	>250	Alive
Tlatlos #2	COG	WH	450	74.6	21.9	36.3	>250	Dead
Tlatlos #3	COG	WH	460	84.0	24.3	44.5	>250	Alive
Tsitika	COG	WH	520	79.4	21.9	72.4	>250	Alive
Patterson Lake #1	CSG	DF	280	69.9	18.5	35.4	60–80	Alive
Patterson Lake #2	CSG	WH	300	72.5	10.6	30.2	60–80	Alive
Patterson Lake #3	CSG	WH	285	46.0	16.5	35.6	60–80	Alive
Patterson Lake #4	CSG	DF	370	51.5	17.3	40.7	60–80	Alive
Cervus Creek #1	COG	DF	423	84.5	19.3	64.3	>250	Alive
Cervus Creek #2	COG	DF	440	92.5	41.0	112.0	>250	Alive
Cervus Creek #3	COG	DF	430	82.5	21.5	53.6	>250	Alive
Cervus Creek #4	COG	DF	ND	ND	ND	ND	>250	Alive
Anderson Lake #1	COG	WH	230	68.0	13.8	51.6	>250	Alive
Anderson Lake #2	COG	ND	ND	ND	ND	ND	ND	ND
Cowichan Lake	CSG	DF	260	33.0	6.7	27.0	50–60	Alive
Mt. Edinburgh #1	COG	WH	230	58.7	21.7	35.8	>250	Alive
Mt. Edinburgh #2	COG	ND	ND	ND	ND	ND	ND	ND
Mt. Edinburgh #3	COG	ND	ND	ND	ND	ND	ND	ND
Lupin Falls	COG	DF	475	57.6	26.2	44.5	>250	Alive
Derby #1	COG	WH	420	56.2	19.3	41.3	>250	Alive
Derby #2	COG	WH	430	59.6	17.2	36.5	>250	Alive
Derby #3	COG	WH	490	58.5	11.0	23.0	>250	Alive
Claud Elliot #1	COG	WH	550	52.3	20.3	35.5	>250	Alive
Claud Elliot #2	COG	WH	500	76.0	20.5	27.5	>250	Alive
Lukwa	COG	WH	650	82.6	14.2	42.8	>250	Alive
Klaklakama #1	FRAG	WH	450	89.5	32.7	ND	>250	Alive
Klaklakama #2	FRAG	DF	500	103.0	24.8	56.4	>250	Alive
Klaklakama #3	FRAG	DF	510	104.0	32.0	ND	>250	Alive
Klaklakama #4	FRAG	WC	620	90.1	15.0	31.2	>250	Alive
W. Cracroft Is. #1	CSG	DF	80	31.0	ND	ND	60–80	Dead
W. Cracroft Is. #2	CSG	DF	ND	ND	ND	ND	60–80	Alive
Nimpkish Is. #1	FRAG	DF	200	112.0	46.1	82.4	>250	Alive
Nimpkish Is. #2	FRAG	DF	250	178.0	22.7	63.2	>250	Alive
Iloomak Lake	FRAG	DF	270	93.3	21.7	74.2	>250	Alive
Consort Creek	COG	WH	670	70.7	20.1	37.1	>250	Alive
Pye Lake	CSG	RA	305	25.5	11.9	16.7	60–80	Alive
Upper Stella	CSG	RA	270	37.3	ND	ND	60–80	Alive
Rona Loop	FRAG	DF	170	112.5	20.2	44.8	>250	Alive
Vernon Ridge #1	COG	WH	435	95.3	55.7	79.8	>250	Alive
Vernon Ridge #2	COG	DF	420	100.3	24.8	57.9	>250	Alive
Nahmint #1	COG	DF	450	82.5	25.1	>60	>250	Alive
Nahmint #2	COG	ND	ND	ND	ND	ND	>250	Alive
Museum Ck.	CSG	RA	380	25.0	11.4	20.0	50–60	Alive
Cottonwood #1	CSG	RA	550	ND	ND	ND	50–60	Alive
Cottonwood #2	CSG	RA	550	ND	ND	ND	50–60	Alive

Table 3. Continued.

Nest	Forest type	Tree species[a]	Elevation(m)	DBH[b] (cm)	Nest ht.(m)	Tree ht.(m)	Tree age (yr)	Decay
Cook Creek	CSG	RA	370	22.8	ND	ND	50–60	Alive
Supply Creek	CSG	DF	250	42.2	13.9	30.4	68–70	Alive
Quadra Island	FRAG	SS	220	44.4	12	22.8	74	Dead
China Creek	COG	ND	ND	ND	ND	ND	>250	Alive
Goose Creek	CSG	DF	ND	ND	ND	ND	50–60	Alive
Gold Park	COG	DF	280	99.3	13.8	38.9	>250	Alive
Muchalat Lake	COG	DF	430	42.9	19.4	22.5	>250	Alive
Oktawanch	COG	DF	810	85.9	ND	33.1	>250	Alive
TOTALS, Means±SE	18: COG 11: CSG 6: FRAG	29: DF 19: WH 6: RA 1: SS 1: WC	375±31.4	68.6±5.8	20.5±2.1	42.6±4.4	40: >250 18: 50–80 3: ND	52: Alive 4: Dead 3: ND

[a] DF = Douglas-fir; WH = western hemlock; WC = western redcedar; RA = red alder; SS = Sitka spruce.
[b] DBH = diameter at breast height.

than broadcast surveys and it is limited to areas with good vantage points near forested stands.

Broadcast surveys and stand-watches appear to be more useful during certain times of the summer than others, corresponding to specific periods during goshawk breeding phenology. There seem to be 2 peak times for conducting broadcast surveys: 1 when nestlings are young and adults respond to the alarm call, and another when young are in the late-nestling phase or early-fledgling phase and respond to the juvenile food-begging call. These results are similar to broadcast survey experiments performed by Kennedy and Stahlecker (1993) and Watson et al. (1999). As well, the likelihood of locating a goshawk nest after a detection is greater during the nestling phase because detections are most often within 100 m of nests, whereas during the post-fledgling phase detection rates are frequently greater than 100 m from nests (Kennedy and Stahlecker 1993, Watson et al. 1999). Thus, it is important to determine the dates corresponding to these stages in the breeding phenology of goshawk pairs each year, as they will vary annually and geographically (Squires and Reynolds 1997), so that survey efforts may be concentrated during these time periods.

Stand-watches have been most effective during the late post-fledgling period for detecting the presence of immature goshawks, and therefore, may be useful for supplementing broadcast surveys during this time period. Additionally, stand-watches have proven to be useful during courtship (March and April) to detect the presence of adults within territories (C. Crocker-Bedford, U.S. Forest Service, AK, 1996, pers. comm.; Chytyk et al. 1997).

Two other inventory techniques may have potential for detecting goshawks and locating their nest sites. These include dawn vocalization surveys during the courtship period and broadcasting the "chup" call, a goshawk vocalization given by males when they enter the nest stand with food (Sutton 1925, Schnell 1958, Palmer 1988, Squires and Reynolds 1997) during the courtship, nestling, and post-fledgling periods. Currently, both these techniques are being tested on Vancouver Island. Dawn vocalization surveys are an effective method for locating Cooper's hawk nests in the urban environment of Victoria, B.C. (Stewart et al. 1996) and were 100% effective for detecting goshawks within known nest territories in France (Penteriani 1999). However, it remains unclear how feasible dawn vocalization surveys will be due to the remote and often rugged areas where goshawk nests are situated on Vancouver Island, the limitations of poor weather conditions during the courtship period, and their utility for locating new goshawk territories.

In 1997 and 1998, 2 new goshawk territories were located by tracking a radio-tagged female that moved to nest in a new location from the previous year. Thus, radiotelemetry has proven to be an additional method for locating alternate nests within known territories and for locating new territories. Because field crews can only survey a small portion of Vancouver Island each summer, educational talks to industry and government field workers and to members of the public are also a valuable means of generating reports of new goshawk nest locations.

Territory Occupancy and Nest Productivity

Occupancy rates at known goshawk territories on Vancouver Island have decreased between 1996 and 1998 (years where sufficient data are available). Occupancy averaged 90.0% (n = 11) in 1996, 68.4% (n = 19) in 1997, and 52.0% (n = 25) in 1998 (McClaren 1997, 1999). It is difficult to know whether this decrease in territory occupancy from 1996 to 1998 is merely a function of increased sample sizes or whether there are other influential factors. Occupancy rates are difficult to

compare among studies because rates vary according to the methods used and the amount of effort expended to determine occupancy (Kennedy 1997). However, when occupancy rates are averaged for 1996–1998, mean occupancy on Vancouver Island is 70.1%, which is similar to other studies (New Mexico: 74.4%, n = 22; Utah: 74.7%, n = 26, Kennedy 1997; California: 74%, n = 26, Woodbridge and Detrich 1994). It is not unusual for occupancy rates to fluctuate annually for goshawks (Kennedy 1997). Consequently, the low rates observed in 1998 may just reflect a year with low numbers of nesting goshawks rather than indicating a decreasing trend in occupancy rates for goshawk territories on Vancouver Island.

Although territory occupancy was relatively low in 1998, nest productivity was relatively high, with an average of 1.9±0.2 fledglings per active nest (including newly found and previously known territories). This is higher than in 1996 and 1997 when nest productivity averaged 1.7±0.2 (n = 17) and 1.4±0.2 (n = 17), respectively (McClaren 1997). Including newly found nests in nest productivity estimates has been shown to inflate productivity values because nests that fail are less likely to be found and are therefore not included in productivity estimates (Steenhof 1987). However, with this in mind, mean nest productivity for Vancouver Island appears to be similar to other studies where productivity values range from 0.0 to 2.8 (Doyle and Smith 1994). Furthermore, the number of fledglings per nest ranged from 1.0 to 2.1 in Oregon (DeStephano et al. 1994), averaged 1.8 in Washington (Sheets 1993), and averaged 2.3±0.4 in Mediterranean Italy (Penteriani 1997).

Annual fluctuations in territory occupancy and nest productivity are hypothesized to be caused by changes in food supply (Doyle and Smith 1994, Ward and Kennedy 1996), and weather (Penteriani 1997). Newton (1979) documented that raptors will postpone breeding when food resources are limiting. For example, goshawk reproduction in the Yukon has been demonstrated to cycle with the abundance of their primary food source, snowshoe hare (*Lepus americanus*; Doyle and Smith 1994). As there are no snowshoe hare on Vancouver Island, red squirrel (*Tamiasciurus hudsonicus*) rather than snowshoe hare, are considered to be an important prey item for goshawks (T. Ethier, B.C. Ministry of Environment, Lands and Parks, Penticton, 1995, pers. comm.). However, a relationship between red squirrel numbers and goshawk nest success remains to be demonstrated.

In terms of weather conditions, research on goshawks by Penteriani (1997) in the Mediterranean demonstrated goshawk nesting success to be closely correlated to rainfall levels during the spring and, particularly, during the incubation period. During years of cold, wet springs Penteriani (1997) found that goshawks began incubation later and suffered reduced nesting success, possibly due to decreased hunting and food intake. This aspect has to be further examined for goshawks on Vancouver Island, but high snowfall levels and occasional extremely wet months of March, April, and May could be factors contributing to years of lower nesting success and reduced nest productivity. Nest predation can also be a factor that reduces the number of fledglings at nest sites (Ward and Kennedy 1996); however, we have no evidence of higher than average nest predation rates on Vancouver Island.

Some researchers have suggested that certain habitat characteristics influence territory occupancy and nest productivity and that over the long run, habitat fragmentation and removal may have a negative impact on goshawks (Woodbridge and Detrich 1994, Patla 1997, Desimone 1997). Territory occupancy has been demonstrated to be positively correlated with: nest stand and nest stand cluster size (defined as the aggregate of all nest stands within one pair's territory; California: Woodbridge and Detrich 1994); the amount of mid-aged, closed forest (average stand DBH: 23–53 cm, >50% canopy cover) and late, closed forest (>53 cm DBH, >50% canopy cover) at 5 spatial scales (12, 24, 52, 120, and 170 ha; Oregon: Desimone 1997); and significantly more mature forest cover (DBH: 45.7–60.9 cm) within the nest area (81 ha) and post-fledgling family area (162 ha), less seedling (DBH: 2.54–12.7 cm) and young forest (DBH: 12.7–30.5 cm) cover, and with the proportion of sagebrush/shrub within 2,428 ha (eastern Idaho; western Wyoming: Patla 1997). Although nest productivity was not influenced by stand cluster size in California (Woodbridge and Detrich 1994), it was positively related to basal area at the 81-ha scale and to the proportion of sagebrush/shrub at the 2,428-ha scale in eastern Idaho/western Wyoming (Patla 1997). Regression analysis failed to show a significant relationship between forest age class and pooled occupancy rates or nest productivity in Patla's (1997) research, most likely because a large portion (mean of 60%) of territories were located in mature forest, so there were few habitat differences among nest sites.

On Vancouver Island, where there are dramatic habitat differences among nest sites, territory occupancy has been consistently higher in continuous old-growth forests compared to continuous second-growth and fragmented forests. Nest productivity, on the other hand, appears to be similar between continuous old-growth and continuous second-growth forests, although it is slightly lower in fragmented forests. Territory occupancy and nest productivity in relation to forest age class and the amount of fragmentation within goshawk home ranges is being further investigated (McClaren in prog.).

Nesting Habitat

Within the literature there exists a great deal of debate regarding the habitat requirements of goshawks (Clark 1998, Daw et al. 1998). Most habitat information has been collected

around nest sites during the breeding season, whereas data on the winter habitat associations of goshawks are scant (Kennedy 1997, Squires and Reynolds 1997). In a status evaluation of goshawks, the United States Fish and Wildlife Service found that goshawks, although typically nesting in mature or old-growth forests, appear to be able to use many ages and types of forests to fulfill their life history requirements (Clark 1998). Conversely, a literature review of the habitat characteristics around goshawk nests in the western United States by Daw et al. (1998) concluded that, regardless of region or forest type, there was a trend for goshawks to nest in stands comprised of large trees (>53 cm DBH) with high canopy closure (>50–60%). To further complicate matters, biased sampling of goshawk breeding habitat in many studies may have misconstrued our understanding of their habitat associations (Kennedy 1997, Daw et al. 1998, Rosenfield et al. 1998).

More in-depth descriptions of habitat associations of goshawks on Vancouver Island at the nest site, stand, and landscape levels are described by Ethier (1999). From general habitat data collected at goshawk nest trees during inventory efforts, goshawks appear to nest in a variety of tree species on Vancouver Island, including Douglas-fir, western hemlock, Sitka spruce, red alder, and western redcedar. Although most nests are in Douglas-fir and western hemlock, many nests in second-growth forests are in red alder trees. Thus, it appears that goshawks select for tree structure rather than species, and if trees have sufficient support structures for goshawk nests and are surrounded by forests that will provide forage opportunities, protection from predators, and suitable habitat to raise young, (i.e., high canopy closure and large DBH; Daw et al. 1998), goshawks will nest in these locations.

More recently, the importance of understanding goshawk habitat relationships at larger scales has been acknowledged (Kenward 1982, Bosakowski and Speiser 1994, Bright-Smith and Mannan 1994, Hargis et al. 1994, Squires and Reynolds 1997, Widen 1997, Daw et al. 1998). However, there is a paucity of information available at this scale. The results of goshawk habitat studies at larger spatial scales are equivocal. Some investigators claim that goshawks require relatively large, undisturbed tracts of older forest to nest successfully (Bosakowski and Speiser 1994, Desimone 1997, Patla 1997). Alternatively, others suggest that goshawks are able to nest in small (0.4–20 ha), widely spaced stands of mature forest (Lindell 1984, Woodbridge and Detrich 1994, Younk and Bechard 1994, Squires and Ruggerio 1996, Widen 1997). Widen (1997) and Beier and Drennan (1997) propose that fragmentation of foraging habitat as it relates to prey availability and abundance may be more limiting to goshawks than suitable nest sites. In fact, Widen (1997) suggests that goshawk populations in Sweden, Finland, and Norway (Fennoscandia) are declining due to the deterioration of goshawk foraging habitat in the boreal forests and that this decline cannot be overcome by creating protected areas, because they need areas that are much too large to be effectively protected in this manner. Thus, whether goshawks are forest generalists at large spatial scales, as some researchers claim (Reynolds et al. 1992, Squires and Reynolds 1997, Daw et al. 1998), and the types and amount of habitat they require to meet their life history requirements, remain unanswered.

At a larger scale, goshawk nests on Vancouver Island are generally located on the bottom to middle portion of forested slopes, at lower elevations, on moderate slopes. After 2 years of goshawk inventory on Vancouver Island, the majority of goshawk nests were found on slopes with southwest aspects (Quayle et al. 1995). Therefore, from these data it was assumed that goshawks on Vancouver Island preferred to nest on forested slopes with southwest aspects. However, there were only 11 nests known at this time. Since 1995, goshawk nests have been found on all aspects and there is no particular trend in nest stand aspect. In the southern United States, goshawk nests are typically found on north- and east-facing slopes (Reynolds et al. 1982, Hayward 1983, Bull and Hohmann 1994, Crocker-Bedford 1994). In southeast Alaska, Titus et al. (1994) reported most nests to be on northeast aspects, whereas McGowan (1975) described the majority of goshawk nests to be on southern slopes. Elevation and aspect may be important habitat selection criteria for goshawks, as these factors influence the microclimate around their nests. However, there has been minimal information collected in this area of goshawk ecology (Squires and Reynolds 1997). If microclimate is important to goshawk nesting success, we would expect nests in southern latitudes to be on cooler, northeast aspects to prevent overheating, whereas nests in northern latitudes should be on the warmer, southwest aspects. Although data from the southern states support this hypothesis, data from Vancouver Island and southeast Alaska suggest that other habitat characteristics (i.e., tree density, DBH, canopy closure) are more influential in determining where goshawks establish nest territories.

Furthermore, data collected from Vancouver Island do not suggest that goshawk nest locations are associated with particular elevations, as this may also influence the heat regime around goshawk nest sites. Goshawks have been found nesting from sea level in southeast Alaska to high-elevation forests in Colorado (Squires and Reynolds 1997). On Vancouver Island, goshawk nests range in elevation from 80 to 810 m, with most nests located between 300 and 500 m. Nests are typically below 900 m on Vancouver Island and in southeast Alaska (Titus et al. 1994), probably because forest stand characteristics change at higher elevations. Above 1,000 m, most forests on the east coast and central parts of Vancouver Island change from being dominated by western hemlock, Douglas-fir, western redcedar, and amabalis fir

(*Abies amabalis*) to domination by mountain hemlock (*Tsuga mertensiana*), yellow-cedar (*Chamaecyparis nootkatensis*), and subalpine fir (*Abies lasiocarpa*), with a dense salal (*Gaultheria shallon*) understory. Additionally, forests at higher elevations have more open canopies and smaller trees. Therefore, it appears goshawks on Vancouver Island, and possibly in southeast Alaska, are restricted to nesting below 1,000 m. However, these data may reflect a biased search effort, with most goshawk surveys occurring in the more easily accessed, lower elevation forests.

CONCLUSIONS

In conclusion, goshawk inventory has occurred for 5 years within the same areas on northern Vancouver Island. Repeated inventory on northern Vancouver Island continues to provide new and interesting information about goshawks that enables us to adapt habitat suitability models and manage more effectively for goshawks in British Columbia. Additionally, expanding goshawk inventory into new areas of Vancouver Island each year allows us to gather more information on their distribution and habitat associations. Although a valuable baseline of information was gathered by Ethier (1999) on goshawk nest site selection, nest stand structure, and prey use, there are still several information gaps for goshawk ecology on Vancouver Island that must be filled. These include: population size estimates; landscape-level habitat requirements; breeding and winter home range size estimates; juvenile and adult survivorship; and juvenile dispersal distances. This information is critical so that we can gain a better understanding of the goshawk's role in the forest ecosystems on Vancouver Island and learn more about how goshawks respond to current logging practices. Management guidelines outlined in the *Managing Identified Wildlife: Procedures and Measures, volume I* (B.C. MOF and B.C. MELP 1999) for goshawks in British Columbia were primarily developed from goshawk studies conducted outside British Columbia, due to the lack of local information on goshawk populations. This emphasizes the importance of continuing goshawk inventory and research on Vancouver Island and throughout the rest of British Columbia. We must guarantee that science-based practices are used to manage goshawk populations appropriately to ensure populations persist into the future.

ACKNOWLEDGEMENTS

Financial support from Forest Renewal British Columbia and the Habitat Conservation Trust Fund and the hard work of several field members have made goshawk inventory and research on Vancouver Island possible between 1994 and 1998.

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Dispersal of Garry Oak Acorns by Steller's Jays

Marilyn A. Fuchs[1]

Centre for Applied Conservation Biology
University of British Columbia, Forest Sciences Centre
4041-2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada

Pam G. Krannitz

Canadian Wildlife Service
R.R. 1, 5421 Robertson Road, Delta, BC, V4K 3N2, Canada

Alton S. Harestad

Department of Biological Sciences, Simon Fraser University,
Burnaby, BC, V5A 1S6, Canada

ABSTRACT

Acorns of Garry oaks (*Quercus garryana*) in British Columbia are dispersed primarily by Steller's jays (*Cyanocitta stelleri*). We conducted a planting experiment and observed caching by Steller's jays to examine factors influencing acorn and seedling survival and to assess the role of the jays in oak regeneration. Almost all acorns cached by jays were hidden beneath ground cover or soil and thus protected from predation and desiccation. Steller's jays preferred to cache acorns in habitats with abundant tree and shrub cover but sparse herb cover. Approximately 50% of the acorns were cached in these habitats, but seedling establishment was low because of acorn predation. The rest of the acorns were cached in other habitats. Some of these habitats provided conditions favourable for oak regeneration. Conservation of Garry oak ecosystems must incorporate strategies that will retain dispersal and regeneration processes.

Key words: acorn, *Cyanocitta stelleri*, food hoarding, Garry oak, *Quercus garryana*, seed dispersal, seedling, Steller's jay.

Garry oak ecosystems are among the most endangered in Canada. Agricultural and urban development coupled with introductions of exotic plant species such as Scotch broom (*Cytisus scoparius*) and orchard grass (*Dactylis glomerata*) have contributed to the destruction, degradation, and fragmentation of these ecosystems in British Columbia (Ussery 1997). Furthermore, fire exclusion threatens Garry oak ecosystems in British Columbia as it does throughout most of the range of Garry oaks (Agee 1993). Prior to European settlement, Coast Salish peoples of southern Vancouver Island burned Garry oak ecosystems regularly to enhance productivity of camas (*Camassia quamash* and *C. leichtlinii*), a nutritious bulb that comprised one of their staple foods (Turner 1991). These fires maintained the open vegetation structure of Garry oak savannahs. Without fire, conifers and other woody species encroach into oak savannahs.

Changes to the extent, species composition, vegetation structure, and disturbance regime of Garry oak ecosystems can affect oak regeneration. As a result, there is growing concern about the adequacy of regeneration of Garry oaks in British Columbia (Erickson 1996). Garry oaks in British Columbia rely upon Steller's jays for acorn dispersal. The jays bury acorns singly in the ground to use as overwinter food, but may not recover all of the acorns they bury, thereby dispersing seeds to caching locations. Regeneration must therefore rely on where and how acorns are dispersed by the jays.

Our objectives were to examine factors influencing acorn and seedling survival and to assess the role of Steller's jays in regeneration of Garry oaks. We examined the effects of acorn burial depth and habitat type on Garry oak regeneration. Both factors are influenced by food-hoarding animals and may be important in Garry oak reproduction. In addition, we determined burial depths and habitats used by Steller's jays to cache acorns and assessed the implications of jay behaviour to reproduction of Garry oaks.

STUDY AREA

Our research was conducted in Metchosin on the southern tip of Vancouver Island, British Columbia. Two sites were used, Mary Hill and Rocky Point, about 2 km apart. Each site is a mosaic of patches of different habitat types. Most of Mary Hill is a south-facing slope. Much of the site has thin soil and prominent rocky outcrops. The oak cover is primarily in savannah form. Rocky Point is relatively flat, and has rocky

[1] Present address: 1461 May Street, Victoria, BC, V8S 1C2, Canada. foxtrec@islandnet.com

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

outcrops as well as areas with deeper soil. Most of the oaks at Rocky Point comprise a woodland.

METHODS

Planting Experiment

We collected acorns from productive oak trees scattered throughout the study sites in September 1994. We selected undamaged acorns and planted them between 10 and 17 October 1994. We planted 2,700 acorns in a split-plot, completely randomized design to test the effects of burial depth and habitat on acorn and seedling mortality. We used 6 replicates of each habitat type at each site. Although most habitat types occurred at both sites, there were a few differences. "Small clump" habitat was present only at Mary Hill. Two habitats, "conifer sapling" and "riparian," were present only at Rocky Point. Ten acorns were planted at each of 3 planting depths within each planting plot: on the surface, under ground cover (leaf litter, matted grass roots, moss, or lichens), and buried in soil just below the soil surface. We visually estimated vegetation cover within a 5-m radius from the centre of each planting plot. Total percent cover was estimated for 3 vegetation layers: herb (0–1 m high), shrub (>1–5 m high) and tree (>5 m high). Plots were revisited in June 1995 and 1996 to determine presence of acorns and seedlings, and condition of acorns and seedlings if they were present.

Data were analyzed using analysis of variance (ANOVA), with habitat and depth as the 2 treatment factors. Values were calculated as proportions and arcsine squareroot transformed to improve normality. For depth comparisons, significant *F*-tests were followed by multiple comparisons of means with sequential Bonferroni adjustments (Rice 1989). For habitat comparisons, significant *F*-tests were followed by Tukey's honestly significant difference tests, which are more powerful than the Bonferroni method when comparing numerous means (SPSS 1997).

Observations of Steller's Jays

We observed Steller's jays from August to October 1994. Each field day, 2 observers walked through different areas of Mary Hill or Rocky Point in search of hoarding activity by Steller's jays. We noted the transport distance of all observations of acorn transport by jays. For each observation of acorn caching, the observer waited until the jay departed and attempted to locate the cached acorn. If the acorn was located, burial depth of the acorn was recorded and the acorn was subsequently replaced. We classified the habitat in which each acorn was cached and estimated total percent cover of vegetation in the herb, shrub, and tree layers for circular plots (5-m radius) centred at the cache location.

We used chi-square tests of homogeneity to compare use and availability of habitat types. If chi-square tests were significant, we calculated confidence intervals around the differences between use and availability (Marcum and Loftsgaarden 1980), with sequential Bonferroni adjustments to control experiment-wise Type I error (Rice 1989).

Table 1. Effects of planting depth and habitat type on percentages of Garry oak acorns removed by predators at Mary Hill in Metchosin, British Columbia, 1995.

Planting depth[a]	Mean percent removal
Surface	89.5A[b]
Ground cover	60.7B
Soil	44.6C
Habitat type[c]	
Small clump	83.9A[b]
Conifer canopy	83.9A
Shrub	77.8AB
Conifer edge	59.4AB
Oak canopy	50.5B
Oak edge	49.4B
Herb	49.4B

[a] $F_{2,70} = 90.72$, $P < 0.01$.
[b] Within planting depth or habitat types, means with the same letters are not different ($P < 0.05$).
[c] $F_{6,28} = 5.47$, $P < 0.01$.

RESULTS

Planting Experiment

Predation on planted acorns was the primary cause of reproductive failure in our experiment. Predation was extremely high for acorns planted on the surface at Mary Hill (Table 1). Deeper planting significantly lowered predation across habitat types. Predation also varied among habitats at Mary Hill. Our experimental design did not permit us to identify which vertebrate species were removing the acorns.

At Rocky Point, an interaction between habitat and depth precluded analysis of depth effect across habitats and habitat effect across depths. Depth effects within each habitat were less consistent than those at Mary Hill, but trends were similar. Predation varied among habitats within each depth (ANOVA and Tukey tests, $P < 0.05$).

Vegetation structure of habitats with low predation at both sites varied, as did vegetation structure of habitats with high predation. All 3 habitats characterized by sparse herb cover, dense shrub cover, and moderate to high tree cover ("small clump" at Mary Hill; "conifer sapling" and "riparian" at Rocky Point) were among those with high predation.

Of the acorns that were not predated, many of those on the surface died (70% and 68% at Mary Hill and Rocky Point respectively), but buried acorns had less than 15% mortality at both sites. Causes of acorn mortality could not be identified in most cases.

Mortality of seedlings at Mary Hill between 1995 and 1996

Table 2. Habitat selection by Steller's jays for caching of Garry oak acorns at 2 sites in Metchosin, British Columbia, 1994.[a]

Habitat type	Mary Hill[b]	Rocky Point[c]
Small clump[d]	+	
Riparian[d]		+
Conifer sapling[d]		+
Conifer edge	0	0
Oak edge	0	0
Mixed canopy	0	0
Mixed edge	0	0
Oak canopy	0	-
Conifer canopy	-	0
Rocky outcrop	-	0
Shrub	-	-
Herb	-	-

[a] preferred = "+" (use > availability, $P < 0.05$); avoided = "-" (use < availability, $P < 0.05$); "0" = no selectivity (use = availability, $P < 0.05$).
[b] n = 179 (use) and 356 (availability).
[c] n = 132 (use) and 409 (availability).
[d] Habitat type present at only 1 site.

varied among habitats (ANOVA and Tukey tests, $P < 0.05$). Seedling desiccation, the only identified cause of seedling mortality, was an important factor in "shrub," "oak canopy," "oak edge," and "herb" habitats at Mary Hill (11–59%). Many of the planting plots in these habitats had thin soil and rocky outcrops, characteristics less prominent in the other habitat types. These habitats are the only ones at Mary Hill lacking a conifer component. Seedling mortality due to undetermined causes was relatively constant among habitats at Mary Hill (<25%). Virtually no desiccation of seedlings occurred at Rocky Point. Seedling mortality was low (<18%) in all habitats at Rocky Point.

Two years after planting, acorns planted in "conifer edge" and "herb" habitats had the highest numbers of seedlings at both sites. ""Small clump" habitat at Mary Hill and "conifer sapling" and "riparian" habitats at Rocky Point had among the lowest numbers of oak seedlings.

Observations of Steller's Jays

Steller's jays transported acorns between a few centimetres and 600 m from the parent tree. At least 1 group of jays appeared to transport acorns more than 1 km. Jays did not transport acorns across a meadow (approximately 150 by 300 m) without scattered trees, but readily transported acorns across open habitats wider than 300 m with numerous scattered trees. The jays frequently used the trees as stopover structures, landing in the trees and surveying the area before continuing on transport journeys. Burial depth of cached acorns was remarkably similar between sites. Virtually all cached acorns were buried. Most acorns (75–76%) were buried in the ground cover layer.

Steller's jays used some habitats to hoard acorns in greater proportion than they were available in the study sites, and used other habitats less than they were available (chi-square and confidence interval tests, $P < 0.05$; Table 2). Steller's jays preferred to cache acorns in habitats with sparse herb vegetation, dense shrub vegetation, and dense cover in the tree layer. These preferred habitats were "small clump" at Mary Hill, and "conifer sapling" and "riparian" at Rocky Point, habitats with high levels of acorn predation. Steller's jays avoided locations with dense herb cover, sparse shrub cover, or sparse tree cover. Despite the high degree of selectivity for the preferred habitat types, only about 50% of the acorns were cached in these habitats. The remainder were distributed among the other habitats.

DISCUSSION

Burial of Garry oak acorns is critical for acorn survival. By burying acorns in the ground cover and soil, Steller's jays protect cached acorns from removal by predators. Although we could not identify the species that were removing the acorns in our experiment, numerous species of birds and mammals consume acorns, many of which were common at Mary Hill and Rocky Point. Burial probably also protects the acorns from desiccation (Borchert et al. 1989, Tietje et al. 1991) although evidence in our study was suggestive rather than conclusive.

Quality of habitat for regeneration of Garry oak was dynamic in our study. Some habitats were good for acorn survival but were poor for seedling survival, and vice versa. Variation in acorn mortality among habitats resulted primarily from predation. Some of this variation was associated with vegetation structure. In contrast, variation in seedling mortality resulted primarily from edaphic and physiographic factors contributing to seedling desiccation. Habitat quality may be similarly dynamic as the oaks continue to develop. In particular, shade intolerance is commonly associated with Garry oaks (Stein 1990, Agee 1993), and evidently becomes more important at a later stage of development.

Habitat preferences of Steller's jays appear to substantially constrain effective dispersal of Garry oaks to favourable sites. Preferred habitats have high rates of acorn predation. Seedling desiccation did not occur in habitats preferred by Steller's jays. However, numbers of acorns removed by predators exceeded numbers of seedlings killed by desiccation, so that seedling numbers in habitats preferred by Steller's jays were among the lowest of all habitats. Furthermore, because of the dense overstory vegetation in preferred habitats, limited light availability probably also limits survival and growth of older Garry oak seedlings or saplings.

Approximately 50% of the acorns were cached in habitats not preferred by Steller's jays. Much of this dispersion resulted when jays landed in patches of preferred habitat but subsequently moved to adjacent patches of different habitat as they

searched for caching sites. Some of these habitats were favourable for oak regeneration. This wide scattering of caches among habitats may be a key component of effective dispersal of Garry oaks by Steller's jays.

MANAGEMENT RECOMMENDATIONS

Conservation of Garry oak ecosystems requires protection not only of extant stands of Garry oak but also management of sites capable of supporting Garry oaks. Ecological processes including dispersal and regeneration of Garry oaks must be retained, and management prescriptions must be guided by an understanding of these processes. Steller's jays transport and cache acorns within, adjacent to, and at long distances from oak stands. Hence, oak regeneration must be considered at both the stand and landscape levels.

To ensure the long-term presence of Garry oak ecosystems in British Columbia, we recommend the following:

1. Efforts to protect remnant patches of Garry oak ecosystems should continue, but adjacent land should also be protected to provide sites for acorn dispersal and seedling establishment.
2. Steller's jays prefer habitats with sparse herb cover, dense shrub cover, and dense tree cover. Habitats with this vegetation structure should be retained to ensure that jays remain part of the ecosystem. Dispersal of acorns to sites favourable for oak regeneration requires that acorns are also transported by Steller's jays to sites with sparse shrub and tree cover. This process is facilitated by the juxtaposition of these habitats with habitats preferred by Steller's jays. Patchy, complex vegetation structure should therefore be retained if present, and creation or restoration of these conditions should be considered.
3. Scattered trees should be planted in open areas to facilitate transport of acorns by Steller's jays.
4. Because Garry oak ecosystems have been severely fragmented and reduced in area, acorns should be planted to establish new oak populations. Planting of acorns just below the soil surface, in habitats on flat terrain with deep soil and sparse overstory vegetation, will ensure greatest survival of acorns and seedlings. Acorns should also be planted in sloping, thin-soiled habitats, where xeric conditions will prevent conifers from establishing and outcompeting slower-growing oaks.
5. Managers must ensure that Steller's jays continue to frequent Garry oak ecosystems in British Columbia. Occupancy of Garry oak ecosystems by Steller's jays is seasonal and our understanding of ecological factors influencing this occupancy is limited (Stewart and Shepard 1994). In particular, continued urbanization may result in declines of Steller's jays in the region.

ACKNOWLEDGEMENTS

We thank M. Arbour, D. Beamish, F. Bunnell, R. Duncan, P. Marshall, L. Reed, and A. Robinson. Field assistants were K. Ferguson and J. Pelletier. The project was funded by the British Columbia Ministry of Forests, Canadian Wildlife Service, Henry P. Kendall Foundation, Natural Sciences and Engineering Research Council, and a Canfor Corporation Fellowship in Forest Wildlife Management. Canadian Forces Base Esquimalt provided access to Mary Hill and Rocky Point and accommodation during the field season.

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Observations of Two Species at Risk in Mainland Southwestern British Columbia: Hutton's Vireo and Western Screech-Owl

Ian Robertson
Robertson Environmental Services Ltd.
1525-200th Street, Langley, BC, V2Z 1W5, Canada
biometrics@bc.sympatico.ca

Martin Gebauer
Enviro-Pacific Consulting
12634-28th Avenue
Surrey, BC, V3A 7P8, Canada

Glenn Ryder
Robertson Environmental Services Ltd.
1525-200th Street
Langley, BC, V2Z 1W5, Canada

Rick Toochin
Robertson Environmental Services Ltd.
1525-200th Street
Langley, BC, V2Z 1W5, Canada

ABSTRACT

Hutton's vireo (*Vireo huttoni huttoni*) and western screech-owl (*Otus kennicottii kennicottii*) are 2 subspecies at risk occupying lowland coniferous or mixed coniferous-deciduous woodlands in the Greater Vancouver and Sunshine Coast areas of British Columbia. Since 1993, we have conducted 43 intensive censuses of the avifauna of 38 sites in this region, including wooded areas proposed for development (84%), and parks and other protected areas (16%). The sampled sites varied from 1.5 to 190 ha, and were all second or third growth, with stand age estimated between 30 and 100 years. We detected Hutton's vireos and western screech-owls at 51% and 42% respectively of all sites censused, comprising 43% and 64% respectively at sites in the Fraser Valley, 54% and 25% on the Vancouver North Shore, and 71% and 29% on the Sunshine Coast. Highest occurrence of both species was in coniferous habitats, followed by mixed coniferous-deciduous habitats. They were absent in deciduous woodland. We conclude that since most survey sites have been completely or partially cleared for land development, the original concerns for Hutton's vireo and western screech-owl populations remain. These concerns may be reduced if studies in the managed forests of southwestern British Columbia confirm the consistent presence of these species in forest stands as young as 25 years old.

Key words: Greater Vancouver, habitat selection, Hutton's vireo, land development, *Otus kennicottii kennicottii*, Sunshine Coast, *Vireo huttoni huttoni*, western screech-owl.

In British Columbia, Hutton's vireo (*Vireo huttoni huttoni*) and the western screech-owl (*Otus kennicottii kennicottii*) share several distinct features. They are frequently a challenge to detect. They have a limited distribution, primarily in low-elevation coniferous or mixed coniferous-deciduous woodlands in the southwestern corner of the province, although the range of the western screech-owl extends eastward where lowland habitats are available (Campbell et al. 1990, 1997). They are both species at risk, particularly in areas facing development pressure, such as the areas within which our surveys took place. Hutton's vireo is Blue-listed (i.e. designated as vulnerable) by the government of British Columbia (Cannings 1998). The *kennicottii* subspecies of the western screech-owl was Blue-listed until it was subdivided into separate subspecies—*saturatus* (Gulf Islands/Vancouver Island) and *kennicottii* (Lower Mainland)—with only the former remaining as Blue-listed (Cannings 1998). On the basis of these ecological and management similarities, we have combined in this paper our observations on these 2 taxonomically distant species.

Owing to the status of these species, they have been a special focus of the avian inventory and environmental assessment projects we have conducted in the past 5 years (1993–98). During this period we have conducted 43 intensive avian inventories of low-elevation woodland sites in the Lower Mainland, on Vancouver's North Shore, and on the Sunshine Coast (Fig. 1). Neither species was a specific target of these surveys, and all sites were surveyed during the day. This explains why in 17 of our surveys we made no special effort to find western screech-owls or their roosts. Though most

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

of our surveys were 1-time, intensive inventories of an area, we made 4 surveys in a North Vancouver park and 3 at a Howe Sound development site to cover multi-seasonal avian use. While the survey locations were nominally opportunistic, we believe their number and distribution encompass representative habitats at risk. The inventories themselves were conducted following systematic survey protocols.

Our principal objectives with this paper have been: to report on the occurrence of these 2 species at the surveyed sites; categorize their occurrence by local geography, habitat, and season; discuss challenges of detection; and comment on habitats at risk for these 2 species in the environments surveyed.

METHODS

Survey methods combined scientific and regulatory requirements. For most inventories (40) we followed the raptor/heron-nest-survey protocol (Barnard n.d., B.C. MELP 1997) because it was a regulatory requirement for development applications. In 1 study area on the Sunshine Coast, where study area stratification was a requirement, we followed forest-and-grassland-songbird-survey methods for the 3 surveys conducted there (B.C. RIC 1997).

In spite of the focus of raptor/heron nest surveys, we used these foot surveys to complete a relatively comprehensive bioinventory in which species richness was the primary objective. The raptor/heron-nest-survey protocol is a line-transect method with a recommended transect interval of 10 m. In practice, we adapted these methods, with approval of regulators, by varying transect width from 10–50 m, depending primarily on visibility and forest structure: 10 m in dense coniferous forest; wider transect widths as the deciduous proportion increased. Search patterns consisted of observers progressing along each of the transects listening for and observing birds (similar to encounter-transect methods discussed below), and plotting the locations of those observed and heard on maps of the study area. Second, in a more active mode, observers mimicked the calls of the northern pygmy-owl (*Glaucidium gnoma*) and added "pishing" sounds to attract certain birds that might have been missed. In those 26 instances where we were searching for western screech-owls as well as Hutton's vireos, the observers closely examined all of the denser thickets, particularly clusters of conifers, for owls or their sign (e.g., guano stains, pellets, feathers), or other indications of an owl roost. For the surveys discussed here, we report on only those western screech-owls that were observed.

At 1 site in Howe Sound (Sunshine Coast), where we sampled on 3 occasions, we used a combination of unlimited radius point counts and encounter transects (B.C. RIC 1997). At each point-count station the observers waited at least 1 minute prior to beginning the count. A count period of

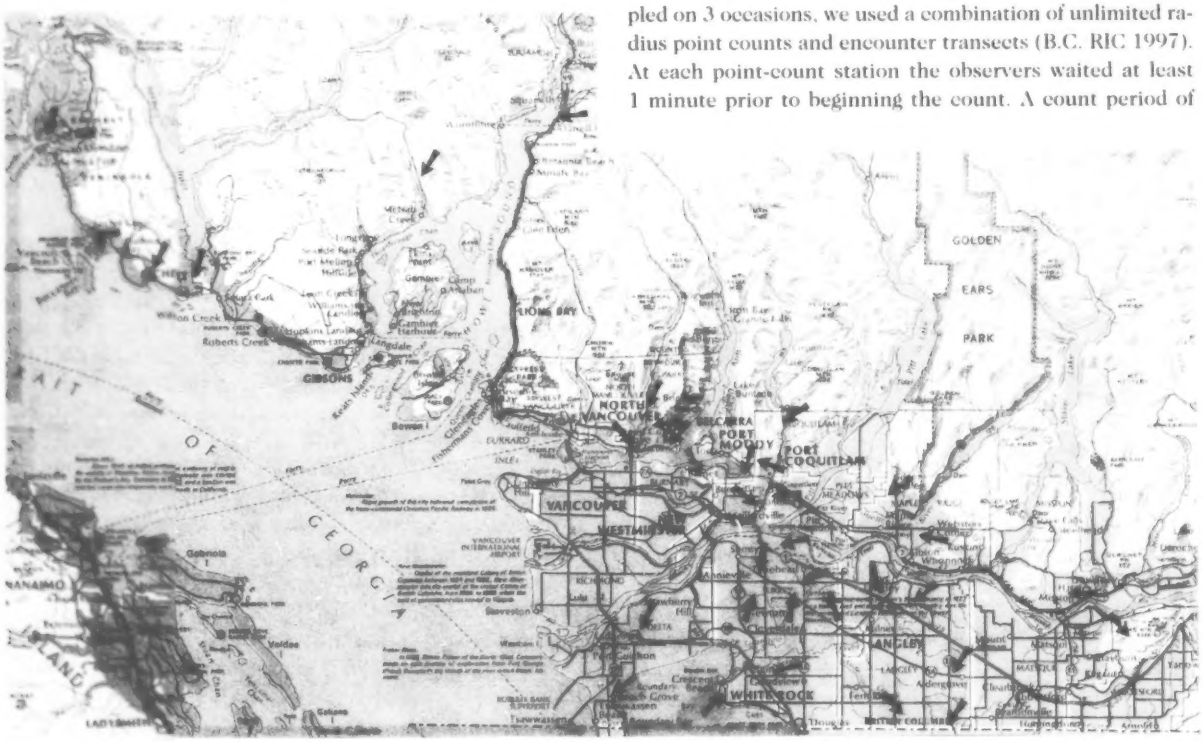

Figure 1. Survey sites for Hutton's vireo and western screech-owl.

Table 1. Size distribution of sampling sites.

Site size (ha)	<5	5–9	10–24	25–49	≥50
No. of sampling sites	9	7	9	11	7

5 minutes was established. During this period, most birds were identified by sound. After individual point counts, we imitated northern pygmy owl calls and began "pishing." Pishing mimics an alarm call of songbirds and, in combination with owl calls, often causes birds that were otherwise undetected to approach the observer and/or vocalize; occasionally these sounds can induce owls to call during daylight. According to British Columbia Resources Inventory Committee (RIC) methodology regarding encounter transects, the observer(s) walks a route and records birds encountered (B.C. RIC 1997). Our encounter transects were a record of all birds observed or heard (mostly) between 1 point-count station and the next. This included the birds recorded in response to simulated owl calls and pishing.

STUDY SITES AND SAMPLING

The choice of sampling sites was opportunistic. The common element was that the sites were being considered for development or were part of conservation areas or parks; in both cases bioinventory was required. The sampling sites were located in the Fraser Valley (54%), at lower elevations of the North Shore (30%), and on the Sunshine Coast (16%). The 43 sampled sites varied in size from 1.5 to 190 ha, with the average sampling area ranging from 25.3 ha in the Fraser Valley to 29.3 ha at Sunshine Coast sites. The distribution of sampling site sizes is indicated in Table 1.

In total, 1,123.5 ha were sampled. While most sites were covered with second- or third-growth forest estimated at 30–100 years old, several sites included small components of shrub habitat or old-field community. On a few sites there was a combination of old-field community, shrub habitat, and several woodland types (Appendix A). Though sampling took place throughout the year, the bulk of it took place between February and July (Table 2).

RESULTS

Frequency of Occurrence

Hutton's vireo

This species was detected at 51% of sites sampled (Table 3). During their February–May breeding season, when their singing is an aid to detection, they were recorded during 12 of 23 samplings. From June through January, when their reduced singing makes them less detectable (Davis 1995), Hutton's vireos were recorded during 10 of 20 samplings. On the basis of a small number of samples, season did not appear to affect detection in this species. The number observed (27) on the Sunshine Coast reflects the 24 Hutton's vireos recorded at a forestry plantation that was sampled on 3 occasions.

Occurrence varied among sites in the Fraser Valley (0.43), the lower elevations of the North Shore (0.54), and the Sunshine Coast (0.71). These results may reflect the higher proportion of coniferous forests and plantations sampled on the Sunshine Coast and North Shore. This is borne out by limited habitat information (Table 4), which is consistent with the apparent preference of Hutton's vireo for coniferous habitats, particularly immature plantations (Bryant et al. 1993, Deal and Lessard 1995).

Western screech-owl

An estimated 889 ha of woodland habitat was surveyed on 26 occasions when we specifically searched for owls at their daytime roosts. We recorded western screech-owls on 42% of those surveys, and the total number of western screech-owls observed was 22 (Table 5). Frequency of occurrence was highest in the Fraser Valley (0.64), with considerably lower occurrence rates on the Sunshine Coast (0.29) and the lower elevations of the North Shore (0.25). Although occurrence

Table 2. Monthly distribution of systematic observations.

Month	J	F	M	A	M	J	J	A	S	O	N	D
No. of surveys	1	8	6	6	3	5	8	0	2	0	1	3

Table 3. Frequency of occurrence of Hutton's vireo.

	All sites	Fraser Valley	North Shore	Sunshine Coast[a]
Area sampled (ha)	1,123.5	582.5	336	205
No. of surveys	43	23	13	7
Frequency of occurrence	0.51	0.43	0.54	0.71
No. of birds observed	59	18	14	27

[a] 5 locations (1 surveyed 3 times)

Table 4. Occurrence of Hutton's vireo by habitat.

	Plantation	Mixed coniferous-deciduous		Deciduous
		>50 yrs	<50 yrs	
No. of surveys[a]	3	21	5	4
Frequency of occurrence	1.0	0.57	0.60	0.00
No. of birds observed	24	23	5	0

[a] 10 surveys are not included because their locations involved a complex mosaic of habitats, which could not be classified

rates were low on the Sunshine Coast, the greatest number of owl observations (11) was from here. All 11 observations were recorded from 1 watershed that flows into Howe Sound; these owls were found in 25- to 50-year-old forestry plantations situated adjacent to the more mature forests that bordered the main creek (Table 6). Other habitats where we found considerable numbers were mixed coniferous-deciduous forests >50 years old. This species was absent in our limited sampling of deciduous woodlands. At individual daytime roosts, western screech-owls were located next to the trunks of western hemlock (50%), western redcedar (40%), and Douglas-fir (10%).

DISCUSSION

Our observations, in combination with those of other researchers, indicate that although Hutton's vireos and western screech-owls are observed frequently and are persisting in the developing areas of the Georgia Depression ecoprovince (Demarchi et al. 1990), the locus of their continued survival could lie in the managed low-elevation forest lands within both the Georgia Depression and those parts of the Coast and Mountains ecoprovince immediately adjacent. With Hutton's vireos, the overall frequency of occurrence (0.51) is broadly comparable with their occurrence during breeding songbird surveys in old-growth and managed forests on western and northern Vancouver Island (Bryant et al. 1993, Deal and Lessard 1995).

Most of the sites we surveyed represented a highly variable combination of second-growth, coniferous-deciduous forest, mostly >50 years old, with a considerable shrub understory, and in which occasionally thick coniferous stands bordered deciduous riparian woodland or edge (clearing, farm field, etc.). This is the model habitat of Hutton's vireo, based on sightings near human settlement in the Georgia Depression ecoprovince (Campbell et al. 1997). At approximately half of all such sites, we found Hutton's vireos. At most of these locations, Hutton's vireos were observed in coniferous trees. The habitat function of deciduous woodland and shrubs, and of edge, for this species is not clear and remains to be determined. There are several possibilities. First, the consistent occurrence of breeding Hutton's vireos in managed forest lands within stand ages from 25 years to old growth may indicate a minor role for deciduous woodland and shrub, at least during the breeding season. Second, in managed forests, it is within Douglas-fir stands that

Table 5. Frequency of occurrence of western screech-owl.

	All sites	Fraser Valley	North Shore	Sunshine Coast[a]
Area sampled	889	475	209	206
No. of surveys	26	11	8	7
Frequency of occurrence	0.51	0.64	0.25	0.29
No. of birds observed	22	8	3	11

[a] 5 locations (1 surveyed 3 times)

Table 6. Occurrence of western screech-owl by habitat.

	Plantation	Mixed coniferous-deciduous		Deciduous
		>50 yrs	<50 yrs	
No. of surveys[a]	3	8	4	2
Frequency of occurrence	1.0	0.62	0.25	0.00
No. of birds observed	11	6	2	0

[a] 9 surveys are not included because their locations involved a complex mosaic of habitats, which could not be classified.

Hutton's vireo occurs; it is virtually absent in western redcedar-dominated forests (A. Bryant, pers. comm.). Since the preferred nursery stock to revegetate the Fraser Valley and southern Vancouver Island woodlands was originally Douglas-fir (R. Merrell, Green Timbers Nursery, pers. comm.), second-growth Douglas-fir in this area is widespread and at an age (>30 years old) that is preferred by Hutton's vireo. Third, the forage provided by deciduous woodlands and shrubs may be vital to the overwintering survival of Hutton's vireo.

Our perspective on western screech-owls was considerably more narrow; we found them primarily at their daytime roosts. Here they were located exclusively in conifers, primarily western hemlock and western redcedar, and in 90% of instances they were pressed up against the tree trunk. This behaviour is consistent with the pattern of their cryptic colouration (Johnsgard 1988). Their relatively high frequency of occurrence in the Fraser Valley and high numbers during 2 surveys at 1 Sunshine Coast location, are similar to habitat occurrence patterns in Hutton's vireo. In the Fraser Valley, the survival of western screech-owls is probably dependent in part on the continued availability of dense coniferous stands within which to roost. The size of such refuges is probably important. We speculate that small woodland refuges (<10 ha) may enhance western screech-owl survival by helping these owls avoid competition with, or predation by, larger owls. Five of the 8 owls observed at Fraser Valley locations were at sites of ≤10 ha. Large owls (great horned owl [*Bubo virginianus*] and barred owl [*Strix varia*]) were recorded during only 2 of the 11 surveys in which western screech-owls were observed. Recent sign of large owls was recorded at 3 additional sites.

The anomalous location among our survey sites was the McNab Creek drainage in Howe Sound. Here, during successive surveys in February and March covering an estimated 50 ha of Douglas-fir plantation aged approximately 25–30 years old and contiguous with mature stands bordering McNab Creek, we found 6 and 5 western screech-owls respectively. (None were recorded when we repeated the survey in June). Their presence in similar numbers was confirmed in nocturnal owl call playback surveys of the same area. The unusual results were related to the high numbers recorded, the very low proportion of deciduous vegetation, and the stand age. This habitat seems far removed from the paradigm of open deciduous and coniferous woodlands bordering riparian corridors (Johnsgard 1988, Campbell et al. 1990), which mostly pre-dated detailed avian studies in managed forests. Somewhat similar results have been recorded by Deal and Lamont (1996) who, in March–May 1995, detected western screech-owls more frequently (0.22) than other owl species in surveys at 353 sampling stations in the Nimpkish Valley on Vancouver Island. Stand age from which responding owls called varied from 15- to 20-year-old regenerating plantations to old growth, plus riparian forest (J. Deal, Canadian Forest Products, pers. comm.). These and similar results from managed forest lands need to be incorporated into our perception of the life history, distribution, abundance, and habitat use of the widening array of avian species of management interest.

Both Hutton's vireos and western screech-owls are easily overlooked. For this reason our surveys probably missed a number of individuals. In addition, our surveys were not directed specifically at these species. As a consequence, certain habitat information was not collected. On the other hand, the raptor/heron-nest-survey protocol that we used in 40 of 43 surveys has no built-in time limit, unlike point counts; thus, our coverage, while somewhat unorthodox, might have been superior to point counts, at least for these 2 species. Survey guidelines and standards recommend experienced observers, but with wildlife management broadening its species focus this is not easy to achieve. The emphasis of our search pattern for Hutton's vireo was first on its vocalizations, particularly its February–May "zu-weep" song, and its rasping "chee-vee-vee" call, variations on which may represent alarm notes uttered throughout the year. Second, we employed "pishing" sounds and mimicked certain avian predators, particularly the northern pygmy owl; this is highly effective in eliciting responses from songbirds and should be stressed. Third, our visual search emphasized the preferred habitats (discussed above) and solitary birds in conifers. With respect to western screech-owls, we do not recommend surveys to find diurnally roosting owls. Other techniques such as owl call playback surveys are equally useful, and have less potential for disturbance.

While our frequency of occurrence data for Hutton's vireo and western screech-owl may seem high for species at risk, it should be remembered that many of the areas surveyed have been partially or completely cleared for land development. Therefore, the original concerns for populations of these species, and particularly Hutton's vireo, remain. New information from certain managed forests indicates that both Hutton's vireo and the western screech-owl are consistently being recorded. If such data are replicated from equivalent studies in other managed forests in southwestern British Columbia, then the status of these species at risk may no longer be valid.

ACKNOWLEDGEMENTS

We thank M. McNicholl and L. Darling for review comments.

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Appendix A. Summary of information from surveys of Hutton's vireo (HUVI) and western screech-owl (WSOW)

Survey no. and date	HUVI	WSOW	Area (ha)	Survey method[a]	Location	Habitat characteristics
1. Sep 1993	0	NS[b]	20	RINS	Mission	Fields, cottonwood plantation, coniferous[c] forest, mixed coniferous/deciduous woodland
2. Apr 1994	3	NS	20	RINS	North Vancouver	Mixed coniferous/deciduous forest (>50 yrs)
3. May 1994	1+	NS	5	RINS	Coquitlam	Mixed coniferous/deciduous forest (>50 yrs)
4. Nov 1994	3	0	23	RINS	North Vancouver	Mixed coniferous/deciduous forest (>50 yrs)
5. Feb 1995	3	1	8	RINS	North Vancouver	Mixed coniferous/deciduous forest (>50 yrs)
6. Feb 1995	4	1	35	RINS	Langley	Mixed deciduous/coniferous woodland (>50 yrs) plus extensive shrubs[c]
7. Apr 1995	0	NS	5	RINS	North Surrey	Mixed coniferous/deciduous forest (>50 yrs)
8. Jul 1995	0	NS	2	RINS	Burns Bog	Mixed coniferous/deciduous forest (>50 yrs)
9. Jul 1995	1+	NS	50	RINS	North Vancouver	Mixed coniferous/deciduous forest (>50 yrs)
10. Dec 1995	0	1	32	RINS	North Surrey	Open fields, mixed deciduous /coniferous[c] forest, coniferous forest, shrub thickets, and small wetlands
11. Feb 1996	0	NS	4	RINS	Walnut Grove	Mixed coniferous/deciduous forest (<50 yrs)
12. Feb 1996	0	0	37	RINS	Coquitlam, Riverview	Deciduous woodland
13. Feb 1996	0	NS	4	RINS	Coquitlam	Mixed coniferous/deciduous forest (>50 yrs)
14. Mar 1996	0	NS	2	RINS	Sumas Mtn.	Mixed coniferous/deciduous forest (>50 yrs)
15. Apr 1996	1+	NS	1.5	RINS	Surrey	Mixed coniferous/deciduous forest (>50 yrs)
16. Apr 1996	1	0	40	RINS	North Vancouver	Shrub, grassy area and playing fields, disturbed area, and mixed coniferous/deciduous forest[c]

Continued...

Appendix A. Continued.

Survey no. and date	HUVI	WSOW	Area (ha)	Survey method[a]	Location	Habitat characteristics
17. Jun 1996	1	NS	50	RIINS	Burnaby	Mixed coniferous/deciduous forest (>50 yrs)
18. Jul 1996	0	0	40	RIINS	North Vancouver	Shrub, grassy area and playing fields, disturbed area, mixed coniferous/deciduous forest [c]
19. Jul 1996	0	NS	15	RIINS	Maple Ridge	Mixed coniferous/deciduous forest (>50 yrs)
20. Sep 1996	0	0	40	RIINS	North Vancouver	Shrub, grassy area and playing fields, disturbed area, and mixed coniferous/deciduous forest [c]
21. Dec 1996	1	1	10	RIINS	Langley	Mixed coniferous/deciduous forest (>50 yrs)
22. Dec 1996	1	2	40	RIINS	North Vancouver	Shrub, grassy area and playing fields, disturbed area, and mixed coniferous/deciduous forest [c]
23. Feb 1997	2	0	20	RIINS	Sunshine Coast	Mixed coniferous/deciduous forest (<50 yrs)
24. Mar 1997	0	1	7	RIINS	Surrey	Mix of deciduous and coniferous woodlands [c]
25. Mar 1997	0	NS	2	RIINS	Langley	Deciduous woodland
26. Apr 1997	1	0	34	RIINS	Langley	Fields, coniferous/deciduous forest, wetland [c]
27. May 1997	0	NS	40	RIINS	North Vancouver	Mixed coniferous/deciduous forest (>50 yrs)
28. May 1997	0	0	8	RIINS	Squamish	Mixed coniferous/deciduous forest (>50 yrs)
29. Jun 1997	0	0	70	RIINS	Tsawwassen	Primarily old field with hedgerows, plus small coniferous/deciduous woodland [c]
30. Jul 1997	1	0	10	RIINS	Port Moody	Mixed coniferous/deciduous forest (<50yrs)
31. Jan 1998	0	0	4	RIINS	Sechelt	Deciduous woodland
32. Feb 1998	1	6	50	FGSS NSOS	Howe Sound	Plantation (25 yrs), mature coniferous, and some mixed coniferous/deciduous woodland
33. Feb 1998	0	0	6	RIINS	Pender Hbr.	Mixed coniferous/deciduous forest (<50 yrs)
34. Mar 1998	2	0	10	RIINS	Coquitlam	Mixed coniferous/deciduous forest (<50 yrs)
35. Mar 1998	0	2	5	RIINS	Surrey	Mixed coniferous/deciduous forest (>50 yrs)
36. Mar 1998	13	5	50	FGSS NSOS	Howe Sound	Plantation (25 yrs), mature coniferous, plus mixed coniferous/deciduous woodland
37. Apr 1998	6	1	190	RIINS	North Surrey	Mixed coniferous/deciduous forest (>50 yrs)
38. Jun 1998	10	0	50	FGSS NSOS	Howe Sound	Plantation (25 yrs), mature coniferous, and some mixed coniferous/deciduous woodland
39. Jun 1998	1	NS	2	RIINS	South Surrey	Mixed coniferous/deciduous forest (>50 yrs)
40. Jun 1998	1	1	45	RIINS	Aldergrove	Mixed coniferous/deciduous forest (>50 yrs)
41. Jul 1998	0	NS	2	RIINS	Maple Ridge	Mixed coniferous/deciduous forest (>50 yrs)
42. Jul 1998	1	0	25	RIINS	Halfmoon Bay	Mixed coniferous/deciduous forest (>50 yrs)
43. Jul 1998	0	NS	10	RIINS	Aldergrove	Deciduous woodland

[a] RIINS = raptor/heron nest survey; FGSS = forest and grassland songbird survey; NSOS = northern spotted owl survey
[b] NS = not surveyed
[c] Habitat not classified for analysis

Coastal Montane Biodiversity Project: Diversity and Structure of the Vertebrate Community

J.B. Joy
598 Millstream Lake Road, Victoria, BC, V9E 1E7, Canada
jeffrey_joy@bc.sympatico.ca

ABSTRACT

Until recently B.C.'s montane forests have remained mostly undisturbed. Submontane timber shortages have led to a dramatic increase in forest harvesting in coastal montane areas. However, little is known about the species and ecosystems present there. Between October 1996 and October 1998 we sampled vertebrate species assemblages as part of the Coastal Montane Biodiversity Project on Mt. Cain and Maquilla peak on Northern Vancouver Island, British Columbia. Our objectives were to provide new information about species presence in the montane, to identify species and communities at risk, and to determine wildlife and habitat relationships in the montane. We confirmed the presence of 65 species of birds, 17 species of mammals, 6 species of amphibian, and 1 species of reptile. Many of these species were previously undocumented in coastal montane habitats, and some occurred only in montane habitats on the coast. Several known rare and endangered species were discovered in the study area. Several unlisted species were documented as rare or in low numbers in these habitats and may be worthy of special management consideration in the coastal montane. We tested species habitat relationships using Linear Regression analysis and Analysis of Variance. Vegetation, elevation, and structural stage appeared to influence coastal montane vertebrate communities on Northern Vancouver Island.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Breeding Bird Censuses in Garry Oak Forests, with Notes on Habitat Use, Metapopulations, and Consequences of Forest Fragmentation

Michael G. Shepard
VGI Vision Group International Inc.
5325 Cordova Bay Road, Suite 211, Victoria, BC, V8Y 2L3, Canada
mgs@vgivision.com

Key words: birds, Garry oak, passerines, *Quercus garryana*, Vancouver Island.

EXPANDED ABSTRACT

During the analysis of data from bird censuses conducted in Garry oak (*Quercus garryana*) forests on southern Vancouver Island, opportunities arose to delve into questions relating to habitat use. This paper discusses possible interpretations of recent observations. Background information is presented in Shepard (1998).

High densities of Pacific-slope flycatchers (*Empidonax difficilis*) found in the study area were surprising. Habitats in the study area are quite arid, and Pacific-slope flycatchers are normally associated with humid forests (Campbell et al. 1997). The patches of oak surrounded by dense clumps of immature Douglas-fir (*Pseudotsuga menziesii*) may be providing a humid microhabitat. I suggest that there are many metapopulations of Pacific-slope flycatchers throughout the species' range, each adapted for surviving in distinctly different habitats. Further studies may show that these populations are disjunct and may, perhaps, be genetically distinct.

Garry oak woodlands having minimal human disturbance are important for cavity nesting birds. In addition to the presence of a moderate number of dead trees, many oaks in these forests have large, dead limbs. This dead wood, with hard shell and soft interior, provides ideal nesting sites. Figure 1 compares the detections of chestnut-backed chickadees (*Parus rufescens*), red-breasted nuthatches (*Sitta canadensis*), brown creepers (*Certhia americana*), and house wrens (*Troglodytes aedon*) in 4 locations on southern Vancouver Island. It indicates that populations are substantially higher in the Garry oak sites (Mary Hill and Rocky Point) than in the Cowichan Valley sites (Currie Creek and Stoltz Road). The latter are primarily second-growth forests of Douglas-fir, with very few trees >60 years old, and very few sites for cavity nesting (Shepard and Manning 1998).

The brown-headed cowbird (*Molothrus ater*) is a brood parasite that can seriously impact songbird populations. Cowbirds prefer open shrubland and forest edge habitats and tend to have the most effect in open areas and forest edges. This trend shows up conspicuously on the 4 southern Vancouver Island sites studied in 1998 (see Fig. 2). The Rocky Point site has an old-growth Douglas-fir overstory, with a broken main canopy of Douglas-fir and Garry oak. Mary Hill is a fairly open site with <50% forest cover; Currie Creek is a second-growth forest that has recently been opened up with small clearcuts (thus providing plenty of edge habitat); and Stoltz Road is basically a forest with a closed canopy. It is suggested that the open sites are more heavily used by cowbirds than densely forested ones. At

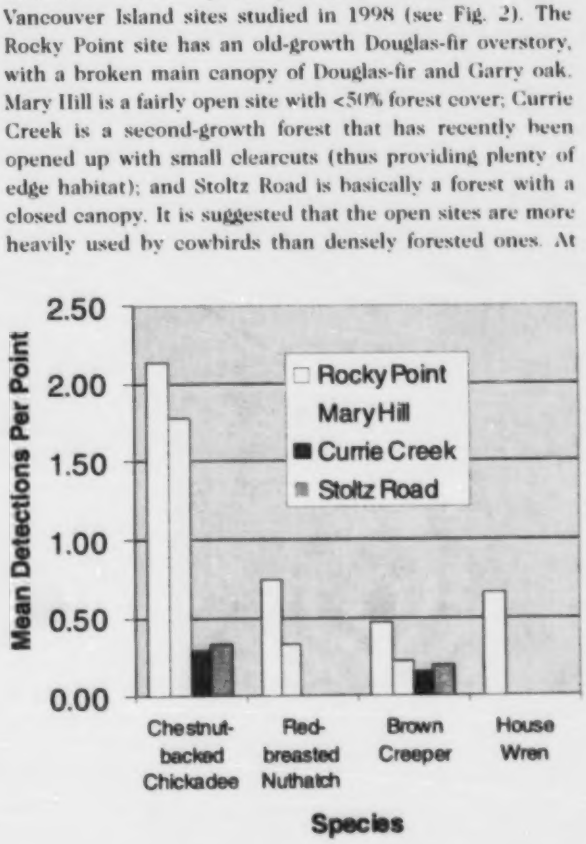

Figure 1. Mean detections of four species of cavity nesting birds at Rocky Point, Mary Hill, Currie Creek and Stoltz Road in 1998.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 498pp.

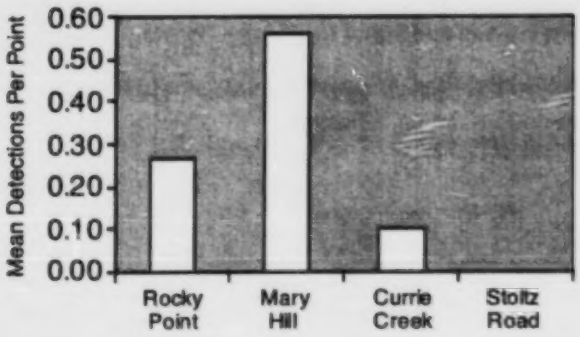

Figure 2. Mean detections of brown-headed cowbirds at Rocky Point, Mary Hill, Currie Creek and Stoltz Road in 1998.

Mary Hill and Rocky Point, many species of birds (golden-crowned kinglet [*Regulus satrapa*], yellow-rumped warbler [*Dendroica coronata*], Townsend's warbler [*Dendroica townsendi*], chipping sparrow [*Spizella passerina*], and dark-eyed junco [*Junco hyemalis*]) were observed feeding fledgling cowbirds.

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The Return of the Purple Martin in British Columbia

David F. Fraser
Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9374, STN PROV GOVT, Victoria, BC, V8W 9M4, Canada
Dave.Fraser@gems8.gov.bc.ca

Darren Copley
Arenaria Research and Interpretation
230 Goward Road, Victoria, BC, V8X 3X3, Canada

J. Cam Finlay
270 Trevlac Place, Victoria, BC, V8X 3X1, Canada

ABSTRACT

In British Columbia, purple martins (*Progne subicolis*) have increased from a few pairs nesting on southern Vancouver Island in the 1980s, to more than 140 pairs on Vancouver Island and the lower mainland in 1998. The increase is attributed to a volunteer-run nest box program initiated in the 1980s and carried on today. Threats to the species' recovery include volunteer "burnout," small numbers, and the decaying condition of pilings on which nest boxes are placed.

Key words: nest boxes, *Progne subis*, purple martin, Vancouver Island.

HISTORY AND STATUS

The purple martin in southwestern British Columbia is regarded as a separate subspecies (*Progne subis arboricola*) from the rest of the Canadian population (*P. s. subis*; Behle 1968, Phillips 1986, Cannings 1998). The *subis* subspecies is only known in British Columbia as a casual wanderer in the northeastern part of the province (Munro and Cowan 1947, Campbell et al. 1997, Cannings 1998).

Historically, the British Columbia purple martin population reached a low of perhaps fewer than 5 pairs in about 1985 (Fraser et al. 1997, Copley et al. 1999). Low population numbers resulted in the species being placed on the provincial Red List (Fraser et al. 1999), and it is a candidate for threatened status under the British Columbia Wildlife Act. The Committee on the Status of Endangered Wildlife in Canada (COSEWIC) has not addressed the *arboricola* subspecies.

Reasons for the decline of purple martins are not known, but it is speculated that they may have experienced considerable competition from introduced species, especially European starlings (*Sturnus vulgaris*; Fraser et al. 1997, Copley et al. 1999), reduction in suitable nest sites associated with modern building construction (Campbell et al. 1997), or that the species may naturally fluctuate in southwestern British Columbia as a result of being at the north edge of its range (Campbell et al. 1997).

Formerly, the species was found on southern and eastern Vancouver Island and the lower mainland (Campbell et al. 1997, Fraser et al. 1997). However, it was extirpated from the lower mainland by the 1950s, leaving only a small breeding population on Vancouver Island (Campbell et al. 1997). Prior to the 1980s, this species nested in natural cavities, holes in buildings, and holes in pilings (Campbell et al. 1997, Fraser et al. 1997).

CURRENT TREND

In the mid-1980s, a nest box program was started for purple martins by members of the Victoria Natural History Society at Cowichan Bay on southern Vancouver Island (Copley et al. 1999). Since that time, expansion of the nest box program has resulted in groups of birds nesting in at least 12 locations in southwestern British Columbia (Fig. 1). In 1998 at least 140 active nests were found (Fig. 2). Breeding was confirmed in the lower mainland in 1994 (Plath 1994), and in 1997 there were at least 14 active nests at Maplewood Flats and Rocky Point, both in North Vancouver (Copley et al. 1999).

The pattern of decline and recovery seen in British Columbia is similar to that seen in Washington and Oregon (Brown 1997, Copley et al. 1999).

CURRENT DISTRIBUTION IN BRITISH COLUMBIA

On Vancouver Island, purple martins are known to breed at Cowichan Bay, Esquimalt Harbour, Ladysmith Harbour, Nanaimo Estuary, Newcastle Island, Nanoose Bay, Campbell

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River Estuary, Oyster Bay, Sooke Harbour, and Victoria's West Bay Marina. On the mainland, they breed at Maplewood Flats and Rocky Point, both in North Vancouver.

CURRENT THREATS

Despite the encouraging trend in numbers, the purple martin is still considered to be at risk in British Columbia (Fraser et al. 1999). The number of individuals is still very small, most of the nest boxes have been placed on decaying pilings, which will disappear over the next decade or so, and maintenance of the next box program is being carried out by a handful of volunteers. More than 90% of the boxes in British Columbia were built and installed and are maintained by a single volunteer! Volunteer "burnout" is a serious threat to the program that maintains this species.

To assist with the expansion of the program, the provincial Wildlife Branch has produced a brochure that outlines the

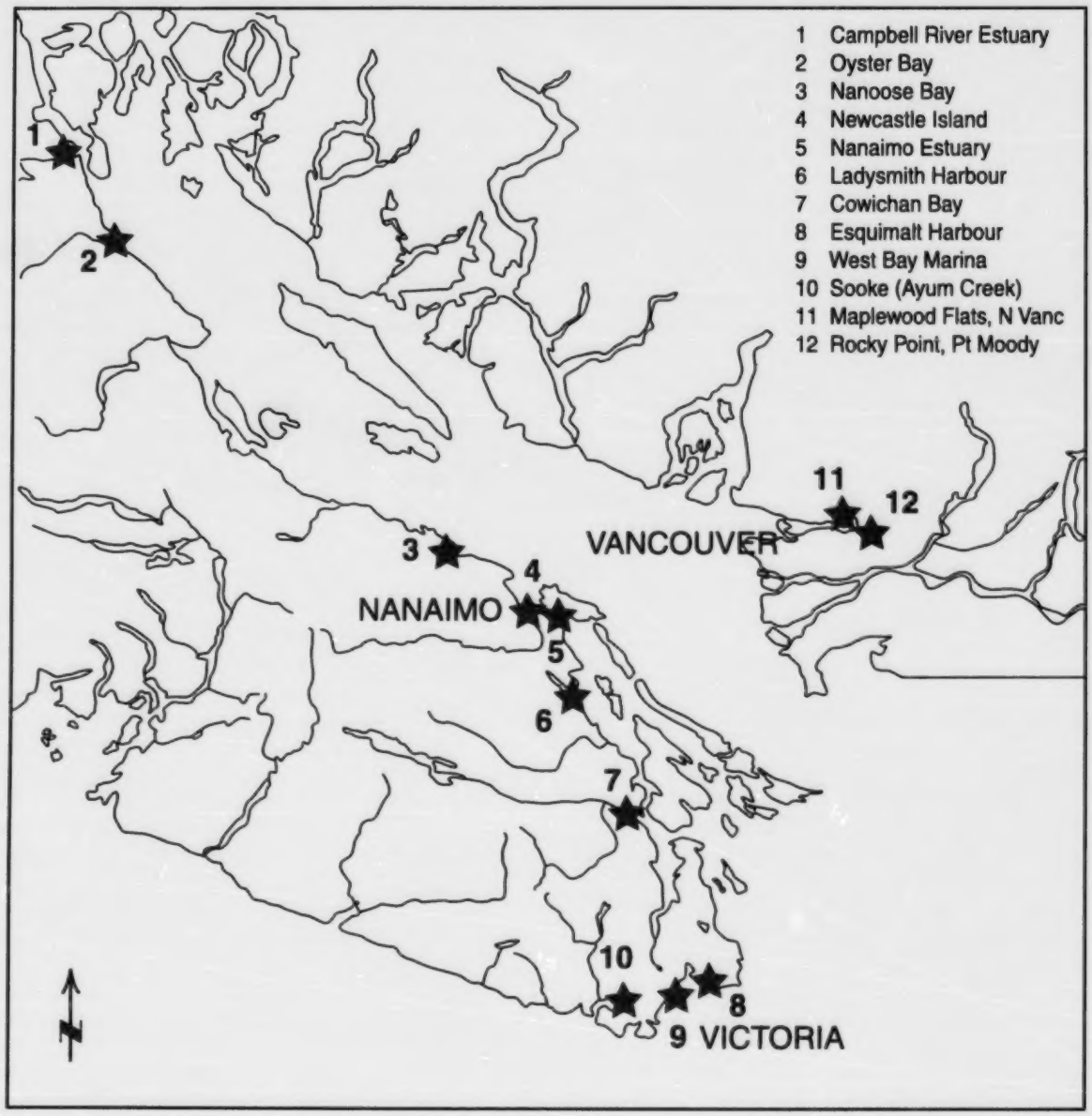

Figure 1. Location of known nest sites of purple martins in British Columbia. Updated from Fraser et al. (1997).

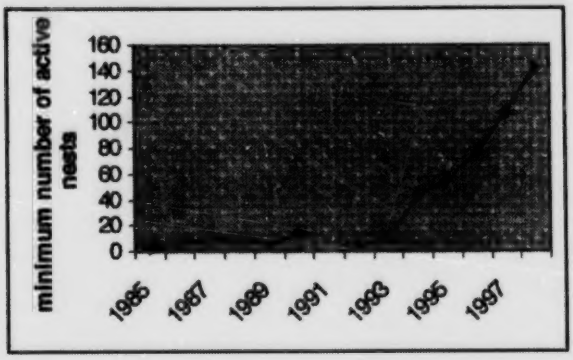

Figure 2. Number of known active nest sites of purple martins in British Columbia from 1985 to 1998. Nest information for 1991 is unavailable. Updated from Copley et al. (1999).

program for purple martin and includes plans for the construction of nest boxes for the species and guidelines for starting new colonies (Pridgeon 1997).

NEXT STEPS

A number of steps for the management of purple martin in British Columbia are outlined in Fraser et al. (1997). At this stage the following are considered to be critical to the maintenance of the species in British Columbia:

- Production of a status report and a listing decision by COSEWIC.
- Development of a recovery and research strategy for the subspecies in Canada.
- Active fostering of stewardship groups to increase the number of nest sites and available nest boxes.
- Acquisition of support for the handful of volunteers that are currently working to recover this subspecies.

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Population Dynamics of Harlequin Ducks in British Columbia and Alberta

Cyndi M. Smith[1]
CWS/NSERC Research Chair in Wildlife Ecology
Department of Biological Sciences, Simon Fraser University
Burnaby, BC, V5A 1S6, Canada

Fred Cooke
CWS/NSERC Research Chair in Wildlife Ecology
Department of Biological Sciences, Simon Fraser University
Burnaby, BC, V5A 1S6, Canada

Gregory J. Robertson[2]
CWS/NSERC Research Chair in Wildlife
Ecology, Department of Biological Sciences
Simon Fraser University
Burnaby, BC, V5A 1S6, Canada

R. Ian Goudie[3]
Canadian Wildlife Service
R.R.1, 5421 Robertson Road
Delta, BC, V4K 3N2, Canada

W. Sean Boyd
Canadian Wildlife Service
R.R.1, 5421 Robertson Road
Delta, BC, V4K 3N2, Canada

ABSTRACT

Harlequin ducks (*Histrionicus histrionicus*) are Yellow-listed in both British Columbia and Alberta, and are listed as a sensitive species in most of the northwestern United States. Moulting and wintering aggregations in the Strait of Georgia include birds known to breed in British Columbia, Alberta, Montana, Idaho, Washington, and Oregon. The wintering population estimate for the Strait of Georgia is 11–15,000 birds, based on capture–mark–recapture studies. Research at White Rock, B.C. shows annual survival rates of 0.82 for adult males, and 0.74 for adult females. Breeding area studies in Banff National Park and Kananaskis Country, Alberta, show that the additional mortality of females takes place during nesting and brood rearing. Breeding ground studies and age ratio counts at the wintering area show that annual productivity and survival of juveniles are both low, resulting in low annual recruitment. These parameters, combined with strong site fidelity at both the breeding and wintering areas, make harlequin ducks susceptible to unnoticed population declines.

Key words: Alberta, Banff National Park, demographic parameters, harlequin duck, *Histrionicus histrionicus*, productivity, Strait of Georgia, survival rate.

The harlequin duck (*Histrionicus histrionicus*) is a sea duck with a Holarctic distribution, and is found on the Pacific and Atlantic coasts of North America. It migrates inland to turbulent mountain streams to breed, the only waterfowl species in the Northern Hemisphere to do so.

The Atlantic population of the harlequin duck was listed as endangered in Canada in 1990, when the numbers were estimated at <1,000 individuals (Goudie 1991). That population appears to be stable or increasing slowly. Best estimates of the Pacific population are 200–300,000, with reductions documented primarily in the eastern and southern parts of the range (Cassirer et al. 1993). In British Columbia, harlequin ducks are considered a species at risk (Lofroth et al. 1996), and they were added to Alberta's Yellow List in October 1996 (Alberta Environmental Protection 1996). The Yellow List is for species for which there has been concern expressed over long-term declines in numbers. The harlequin duck is classified as a sensitive species in Oregon, a priority habitat species in Washington, and a species of special concern in Idaho and Montana (Cassirer et al. 1993).

The Strait of Georgia, B.C. is an important wintering area for harlequin ducks breeding throughout the Pacific Northwest. Marked birds from studies on breeding streams in Alberta, British Columbia, Idaho, Montana, Oregon, and Washington have been re-sighted in the Strait (Fig. 1). Campbell et al. (1990) estimated the wintering population in the Strait of Georgia as >10,000, and recent marking studies put the estimate at 11–15,000 (R. I. Goudie, Harlequin Conservation Society, unpubl. data). With constantly increasing human pressures in the wintering and breeding areas, a better understanding of the demographic parameters of harlequin ducks will enable managers to monitor this important population. The harlequin duck is a good species

[1] Present address: Parks Canada, Box 900, Banff, AB, T0L 0C0, Canada
Cyndi_Smith@pch.gc.ca
[2] Present address: Canadian Wildlife Service, 6 Bruce Street, Mount Pearl, NF, A1N 4T3, Canada
[3] Present address: 17 Waterford Bridge Road, St. John's, NF, A1E 1C5, Canada

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One
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with which to study natural survival rates, as there is little hunting mortality in our study areas.

We will show that harlequin ducks exhibit: 1) high annual adult survival, but higher female mortality in summer; 2) low productivity; and 3) low juvenile survival.

STUDY AREA

Our data come from concurrent work on moulting and wintering areas in the Strait of Georgia, B.C., and on breeding streams in Alberta. The wintering survival study was conducted from July 1994 to November 1998 on a 5.5-km stretch of coastal shoreline that is near the town of White Rock, in southwestern British Columbia. The study on the breeding streams was conducted between May and September (1995–98) on the Bow River in Banff National Park, near Lake Louise, Alberta, and on the Elbow River (1996–98) and Kananaskis River (1998), southeast of Banff. In 1997 and 1998, age ratio surveys were conducted in the Strait of Georgia, between Victoria and Campbell River, during October, November, and December.

METHODS

In the coastal area, moulting harlequin ducks were corralled into a drive trap (Robertson et al. 1998). To capture birds on

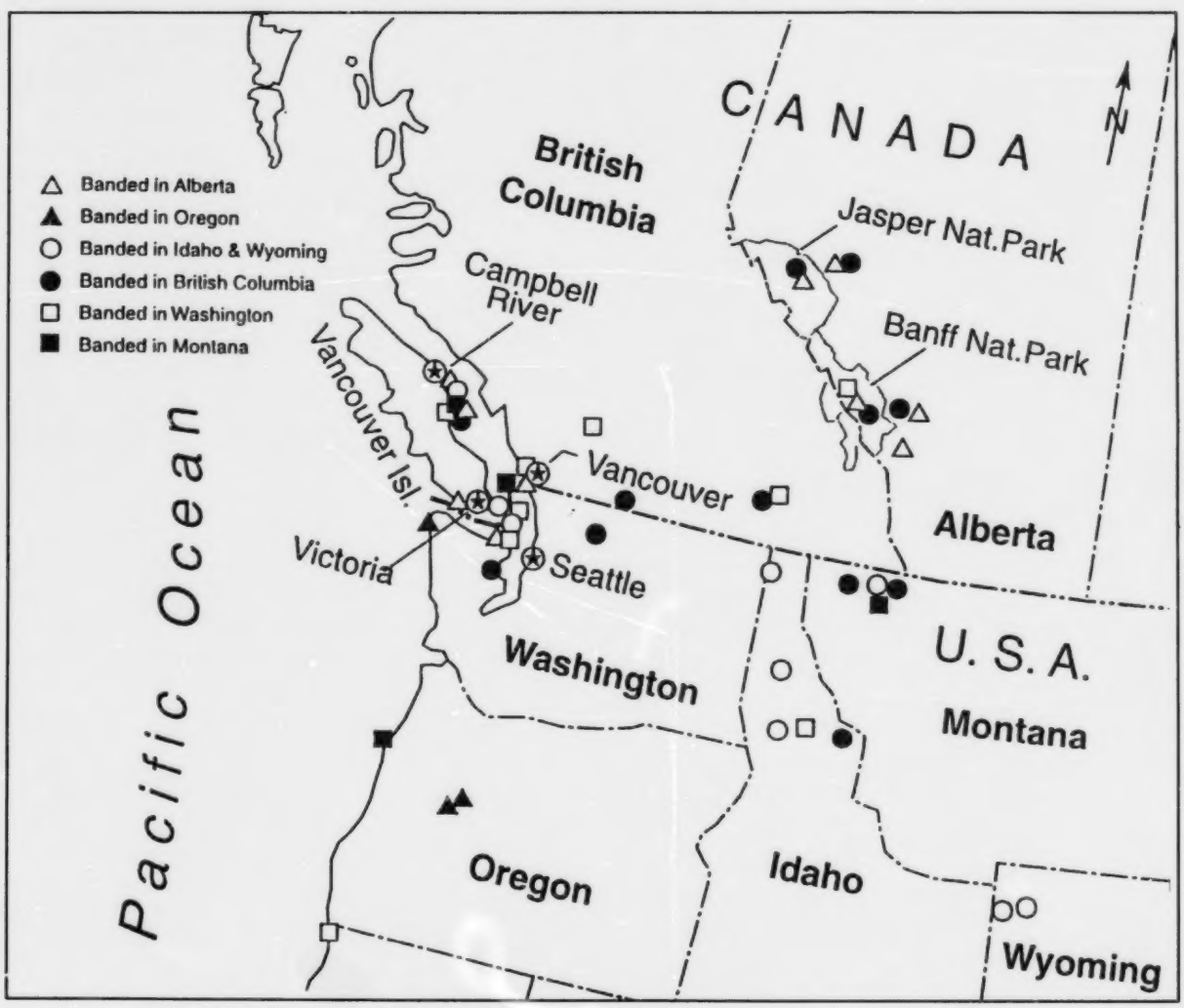

Figure 1. Selected locations of individual harlequin ducks banded on the breeding streams and re-sighted at moulting or wintering areas in western North America, and those banded on the moulting areas and re-sighted on the breeding streams.

Table 1. Survival rates of harlequin ducks in British Columbia (1994–98) and Alberta (1995–98).

Study area	Age and sex	Mean survival rate	Standard error	95% confidence interval
White Rock, BC[a]	adult M	0.822	0.030	0.756–0.873
White Rock, BC[a]	all F	0.739	0.037	0.661–0.805
Banff, AB	adult M and adult F	0.750	0.073	0.582–0.866

[a] Cooke et al. (2000).

the breeding stream, they were actively chased towards a mist-net set across the river (Smith 1996). Captured birds were marked on 1 tarsus with the standard United States Fish and Wildlife Service aluminum band, and on the other tarsus with a coloured plastic band with a unique 2-digit code. Band codes were subsequently read using a 20-45x or 20-60x telescope. Program MARK (White 1999) was used to obtain point estimates and confidence intervals for survival rates.

In addition, radiotelemetry techniques were used on the breeding streams. A subsample of ducks captured in the spring and autumn of 1997, and spring of 1998, were fitted with a 20.1-g intra-abdominal radio-transmitter with an external whip antenna (Korschgen et al. 1996). These transmitters were equipped with a mortality switch that changed pulse after 12 hours of inactivity. Females and young captured in autumn 1998 received a 4.0-g external radio attached mid-dorsally with a prong and sutures (Pietz et al. 1995). These transmitters had temperature thermistors that produced a slower pulse rate as the temperature dropped, indicating a dead bird. This work was performed under Simon Fraser University Animal Care Permit #478B.

During coastal age ratio surveys all harlequin ducks observed were sexed, and males were further classified as either hatch year (HY) or after hatch year (AHY) by plumage characteristics (Smith et al. 1998). Observations were made using 8x binoculars or 20-60x telescopes.

RESULTS

At White Rock, 150 birds were marked with leg bands: 77 males and 73 females, all ages combined. Included were 4 birds that were originally marked in Alberta and which moulted at White Rock. On the breeding streams, 287 birds (97 adult males, 72 adult females, and 118 ducklings) were marked with leg bands; these included 45 adults that were originally marked in other study areas in Alberta, British Columbia, and Washington. Radio-transmitters were attached to 76 birds in 1997 and 1998: 26 females, 7 males, and 43 juveniles.

Adult Survival

Local survival rates for harlequin ducks moulting/wintering at White Rock were 0.822 ± 0.030 (SE) for adult males and 0.739±0.037 for all female age classes (Table 1). These rates are significantly different ($Z = 1.75$, $P = 0.04$). A constant recapture rate of 0.981±0.011 (0.944–0.994) was estimated for the model. Males disappeared from the study area at the same rate in the spring and summer period as in the fall and winter period ($\chi^2 = 0.662$, $P = 0.416$), but females disappeared at twice the rate after the spring and summer period ($\chi^2 = 5.76$, $P = 0.016$; Cooke et al. 2000).

The annual survival rate of the 1995–96 cohort of adult harlequin ducks ($n = 28$) on the Bow River in Banff National Park was 0.750±0.073 for adult males and females to 1998 (Table 1). Of 7 males that received radio implants while on the breeding streams in 1997, none died during their residency on the stream. Of 26 females that received radio implants in 1997 and 1998, 7 died while still at the breeding stream (0.269) for a local summer survival rate of 0.731. Two of the 7 died during the pre-nesting phase (0.077), 1 died during incubation (0.038), and 4 died while accompanying broods (0.154).

Productivity

Twenty of the 26 radioed female harlequin ducks nested (0.769), laid 115 eggs (mean clutch size 6.2), and produced 56 young to class 3 (≥45 days). This was an observed productivity of 2.80 young per breeding female, or 2.15 per total (breeding and non/failed breeding) females. Twenty-three of these ducklings fledged and migrated from the breeding stream, for a real productivity of 1.15 young per breeding female, or 0.88 young per all females.

To make the comparison between productivity observed on the breeding stream to that observed via age ratio surveys we needed to make 2 calculations. We multiplied the 23 ducklings by 0.5 (assumed equal sex ratio of juveniles; Bellrose et al. 1961, Johnson et al. 1992) to arrive at 12 young males, and then multiplied the 26 females by the observed male/female ratio in summer (1.3:1; C. M. Smith unpubl. data) to arrive at 34 adult males on the breeding stream. This gives an estimate of productivity as 0.35 juvenile males per adult male departing the breeding stream.

Age Ratios

The median ratio of hatch-year harlequin duck males ($n = 245$) to adult males ($n = 3305$) was 0.08 (range 0.03 to 0.23) during combined October, November, and December surveys in the Strait of Georgia.

DISCUSSION

For many years wildlife monitoring has involved counting individuals of a species to determine its population size and density. It is increasingly being recognized that these methods are insufficient for conservation purposes (Goss-Custard 1993). Densities alone may be misleading indicators, as they are not necessarily related to habitat quality in a linear manner (Crick et al. 1997). Additionally, for a long-lived species such as the harlequin duck, declines in population size may only be observed after long periods of low survival or reproduction (Crick et al. 1997). Thus, we need to have an understanding of the processes that lead to population changes (Greenwood et al. 1993). These processes are the standard demographic parameters: reproduction, survival, emigration, and immigration. In this paper we attempt to estimate the first 2 processes.

Adult male harlequin ducks had higher local survival rates than adult females. This is consistent with the male-biased sex ratio in many duck species (Bengtson 1972, Sargeant and Raveling 1992). The apparently lower male survival on the breeding stream may not reflect true mortality, as generally only paired males return to the breeding stream. Some males that have only been sighted once may still be alive, but are now breeding in a different area. If a male's original mate dies, he will re-pair and follow his new mate to her natal stream.

During the summer months, adult female survival was lower than adult male survival. As both sexes would be expected to face the same mortality risk during migration, this suggests that higher female mortality is associated with nesting. Incubation and brood rearing are the sole responsibility of the female. Additionally, males are on the breeding streams only 6–8 weeks, while successful females may be there for 16–20 weeks (C. M. Smith unpubl. data), which could increase their predation risk if the breeding stream is riskier than the coastal environment. Winter survival was similar in the 2 sexes, further suggesting that differential mortality of the sexes occurs in the summer.

Barring additional migration mortality, the ratio of juveniles to adults observed at the wintering area should be similar to the ratio of those departing the breeding stream. In our study the median juvenile/adult ratio at the wintering area was 0.08, compared to 0.35 on the breeding stream. This discrepancy could be due to: 1) a higher rate of nonbreeding, or lower productivity, by females in other breeding populations; 2) the majority of juveniles winter in areas apart from adults, and were not covered by the surveys; or 3) there is an even higher mortality of postfledging juveniles on other breeding grounds, or during migration.

The demographic parameters that we have described here—high adult survival rates, low productivity, and low juvenile survival—are typical of sea ducks (Goudie et al. 1994, Krementz et al. 1997). These parameters, combined with strong site fidelity at both the breeding and wintering areas (Smith 1996, Robertson and Cooke 1999), make harlequin ducks susceptible to unnoticed population declines.

ACKNOWLEDGEMENTS

It is difficult to acknowledge the numerous volunteers who have assisted with field captures and re-sightings, but we would like to particularly thank 2 people, K. Wright and B. Gowans. Generous funding has been received from Alberta Environmental Protection, the British Columbia Waterfowl Society, the Canadian Wildlife Service, the Institute for Wetland and Waterfowl Research, Parks Canada, and Simon Fraser University.

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Spotted Owl Management Plan for the Chilliwack and Squamish Forest District

I.R. Blackburn
Ministry of Environment, Lands and Parks, Lower Mainland Region
10470 152nd St., Surrey, BC, V3R 0Y3
Ian.Blackburn@gems1.gov.bc.ca

E. MacInnes
Ministry of Forests, Chilliwack Forest District
Box 159, Rosedale BC, V0X 1X0, Canada

M. Mana
Ministry of Forests, Vancouver Forest Region
2100 Labieux Rd., Nanaimo, BC, V9T 6E9, Canada

O. Thomae
Ministry of Forests, Squamish Forest District
42000 Loggers Lane
Squamish, BC, V0N 3G0, Canada

C.B. Lenihan
Ministry of Environment, Lands and Parks,
Lower Mainland Region
10470 152nd St., Surrey, BC, V3R 0Y3, Canada

R. Thomson
Ministry of Forests, Forest Practices Branch
P.O. 9513, Stn. Prov. Govt.
Victoria, BC, V8W 9C3, Canada

ABSTRACT

The northern spotted owl (*Strix occidentalis*) is designated as Endangered by COSEWIC and is Red-listed by the B.C. Wildlife Branch. In May 1997 the provincial government approved the Spotted Owl Management Plan for the Chilliwack and Squamish forest districts. The long-term management of the species will occur within 21 areas that total approximately 363,000 hectares distributed throughout the range of spotted owls in the Chilliwack and Squamish Forest Districts. This includes 18 Special Resource Management Zones (SRMZs) established under the Forest Practices Code of BC Act. Forest practices within SRMZs will be oriented towards creating, enhancing or maintaining a sufficient quantity and quality of suitable owl habitat. A Resource Management Plan (RMP) for each SRMZ will demonstrate how the objectives of the Spotted Owl Management Plan will be achieved over a long-term planning horizon of one or more forest rotations. Each RMP will identify landscape and stand level management strategies to protect suitable owl habitat and to provide forestry, economic and employment opportunities. The probability of owl population stabilization, and possible improvement, under this plan is about 60%. The short term timber supply impacts associated with the plan are estimated at 3.5 - 4% for the Fraser TSA, 4.5 - 5% for the Soo TSA. Over the long-term timber supply impacts are expected to be reduced to less than 3% in both TSAs. A Spotted Owl Research and Inventory Advisory Committee was established by the provincial government to prioritize research and inventory initiatives that will benefit the adaptive nature of the Spotted Owl Management Plan to improve spotted owl conservation and reduce socio-economic impacts.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

The Northern Spotted Owl: Home Range and Habitat Evaluation

P.R. Mennel
British Columbia Conservation Foundation
Suite 102-15375 102A Ave., Surrey, BC, V3R 7K1, Canada
pmennel@bccf.com

I.R. Blackburn
British Columbia Ministry of Environment, Lands and Parks
Lower Mainland Region
10470 152nd St., Surrey, BC, V3R 0Y3, Canada

ABSTRACT

The Spotted Owl Inventory and Research Advisory Committee (SORIAC) developed a research and inventory strategy in 1997 to support the adaptive nature of the Spotted Owl Management Plan. One component of this plan involves the radio-tracking of 18 spotted owls (*Strix occidentalis caurina*) in southwest British Columbia to determine annual home range size and habitat use. To date 12 spotted owls (5 males and 7 females) have been radio tagged, with nighttime and daytime relocations being utilized to determine nesting, foraging and roosting locations respectively. To determine habitat selection, forest cover data from the British Columbia Ministry of Forests will be tested by ground sampling in the Chilliwack and Squamish forest districts. Characteristics of each type of habitat used will be determined for a coastal and interior ecosystem and possibly a third transitional ecosystem. Based on Washington State habitat classifications, habitats used will be delineated into type A, type B, type C and non-habitat based on their structural characteristics and relative use by the spotted owls. The results will be measured and compared to data from Washington State that is currently being used in the Spotted Owl Management Plan. Findings from this study are expected to: 1) Determine the habitat characteristics of forests used by spotted owls in each ecosystem; 2) Test the assumption that 100 years is the minimum age for Type B suitable habitat; 3) Provide better estimates on the amount of suitable habitat and home range sizes required to maintain viable spotted owl populations in British Columbia; and 4) Be used by the Spotted Owl Management Inter - Agency Team to determine habitat characteristics that may be created or enhanced to increase the probability of the spotted owl population stabilizing in British Columbia.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Natural History of the Sharp-Tailed Snake, *Contia tenuis*, on the Gulf Islands, British Columbia

Christian Engelstoft
P.O. Box 58, Tlell, BC, V0T 1Y0, Canada
alula@islandnet.com

Kristiina Ovaska
Department of Forest Sciences, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada

Key words: *Contia tenuis*, Gulf Islands, sharp-tailed snake, Vancouver Island.

EXPANDED ABSTRACT

"The greatest constraint in conservation planning for either individual species or entire snake assemblages is the fundamental lack of basic biological information on most species" (Dodd 1993, p.370).

The sharp-tailed snake, *Contia tenuis*, is considered endangered nationally by the Committee on the Status of Endangered Wildlife in Canada, and is on the Red List of species at risk in British Columbia. The species is known from only a few sites within the Coastal Douglas-fir biogeoclimatic zone on southeastern Vancouver Island and the Gulf Islands (Spalding 1993, 1995). These populations, which are isolated from the more southern parts of the species' range in the United States, might be unique both genetically and in their ecological adaptations.

During 3 field seasons, from 1996 to 1998, we studied the distribution, life history, and habitat use of the sharp-tailed snake on the southern Gulf Islands and southeastern Vancouver Island (Engelstoft and Ovaska 1997, 1998, 1999). Here, we present a summary and preliminary results of the study with emphasis on natural history of populations at 2 recapture sites on the Gulf Islands: near Magic Lake, North Pender Island (0.75 ha, 48°46'N, 123°18'W), and near Vesuvius, Saltspring Island (0.15 ha, 48°54'N, 123°35'W).

From 1996 to 1998, we searched for snakes for 284 person-hours on the Gulf Islands (North Pender, South Pender, Saltspring, Galiano, Sidney, Saturna, Mayne, Portland, and several smaller islands) and southeastern Vancouver Island (Duncan–Maple Bay area, Shawnigan Lake, Mount Finlayson, and Metchosin) by turning over rocks and logs, looking under bark on decaying logs, and scanning the ground. We found the sharp-tailed snake at 3 localities: near Magic Lake on North Pender, near Vesuvius on Saltspring (at 2 sites approx 1 km apart), and in Metchosin on the southern tip of Vancouver Island. In addition, confirmed observations of the sharp-tailed snake were reported to us from 2 localities, about 2.7 km apart from each other, on South Pender.

All of these sites are at or near previous localities reported for the species. Our observations extend the patch size of habitats occupied by the snakes at the Saltspring and South Pender sites to at least 10 ha and 30 ha, respectively. At the North Pender and Metchosin sites, the snakes are known from areas <1 ha. We were unable to confirm the persistence of the species at a second, previously reported site on North Pender and at 2 historic localities where the exact sites were unknown (Galiano, and Cowichan District, Vancouver Island; Conservation Data Centre, Victoria, files).

Habitats of the sharp-tailed snake consisted of small canopy openings and forest edges in Douglas-fir (*Pseudotsuga menziesii*)/Arbutus (*Arbutus menziesii*)-dominated stands. On the Gulf Islands, the habitats were highly fragmented by residential development and hay fields, whereas in Metchosin, habitat fragmentation was mainly due to small-scale forestry and natural forest openings on exposed bedrock. Small canopy gaps, particularly when along south-facing, rocky slopes, appear to be important for the snakes for thermoregulation, egg development, and growth of young. A 3-dimensional structure of the forest floor, which allows for burrowing or vertical movements, also appears to be required.

In 1997–1998, we caught 24 individuals 1–9 times at the Magic Lake site and 16 individuals 1–5 times at the Vesuvius site. Artificial cover objects, constructed of black asphalt roofing, unpainted tin roofing, and 1.2 cm-thick plywood, allowed us to locate and recapture individual sharp-tailed snakes with minimal disturbance to the natural habitat. Based on the seasonal distribution of captures under artificial covers and rocks, the surface activity of the snakes peaked from March to April in 1997 and from March to April and early June in 1998; a smaller peak in activity occurred from late September to early November in both years. Few snakes were found in July and August, and captures at this time were associated with rainfall. Ten of 21 fecal samples examined contained remains of gastropods, 8 samples contained remains of insects, and 1 sample contained a skeleton of a mite. The availability of prey might partially limit the surface activity of the snakes to moist periods.

The snakes caught ranged from 79 to 323 mm in snout-vent length (SVL). The smallest snakes (79–90 mm SVL)

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were found in September–October, suggesting that hatching occurs at this time. Growth rates for 12 snakes suggest that growth decreases dramatically with body size, being most rapid for first-year young; 1 individual doubled its SVL during its first year of life. The average annual growth rate of 2 larger juveniles <200-mm SVL was 22.4 mm, assuming no growth between November and February. These growth rates would result in SVL of about 200 mm, the body size of most adults, in about 3 years. Nine larger snakes showed slower growth (7.2–13.8 mm/yr).

Movements of individual sharp-tailed snakes, determined from the use of artificial cover objects, were relatively short; the average distance between 2 farthest captures was about 25 m. Six of 18 recaptured snakes were always found under the same cover. In 1997 and 1998, we tracked movements of 2 snakes tagged with subcutaneous implants (weighing approx 4 mg) as part of field trials investigating the usefulness of the harmonic direction finder technique for following movements of small snakes (see Engelstoft et al. in press, for details). The snakes were semi-fossorial and seldom found on the surface. One female snake, detected 57 times from September 1997 to April 1998, moved mainly within decayed logs, under rocks, and underground. The greatest distance between 2 relocation points was 39 m. The longest movement detected for the other tagged snake, an adult male, was 93 m in March–April 1998.

Many aspects of the life history of the sharp-tailed snake, including the age at sexual maturity, timing of mating and oviposition, and frequency of reproduction by females, remain unknown. Seasonal movements and habitat use patterns also require further study. Filling these data gaps is important for management and for the protection of adequate habitats to ensure the persistence of the species in British Columbia.

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Abundance and Movements of Terrestrial Salamanders in Second-Growth Forests of Southwestern British Columbia

Katherine A. Maxcy
Department of Forest Sciences, Forest Sciences Centre
2424 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada
kmaxcy@interchange.ubc.ca

John Richardson
British Columbia Ministry of Environment, Lands and Parks
and Department of Forest Sciences
2424 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

There is a lack of quantitative work on amphibian communities located in second-growth forests in British Columbia. The relative abundance, variation in capture rates, and movement patterns of terrestrial salamanders were investigated in 6 second-growth sites located in the University of British Columbia's Malcolm Knapp Research Forest. These data indicate that while amphibian species richness is similar among sites, total abundance is highly variable both within and among species between sites, with coefficients of variation in capture rates ranging from 44.2% to 245%. This high natural variation in amphibian abundance between forested sites of similar age and structure emphasizes the importance of collecting baseline data before drawing conclusions about the impacts of forest harvesting on amphibians. In addition, aquatic-breeding salamanders (*Ambystoma gracile* and *Taricha granulosa*) moved significantly greater distances and their direction of movement was generally parallel to the stream, while terrestrial-breeding salamanders (*Plethodon vehiculum* and *Ensatina eschscholtzii*) moved less and randomly with respect to stream location. These results suggest breeding biology (i.e., aquatic vs. terrestrial) may be a useful predictor of species' responses to timber harvesting.

Key words: *Ambystoma gracile*, amphibian, *Ensatina eschscholtzii*, forestry, movement, *Plethodon vehiculum*, riparian, salamander, second-growth, *Taricha granulosa*.

The physiological characteristics of amphibians limit them to specific microhabitats and/or microclimates within forest ecosystems; changes to these conditions via timber harvesting may influence population and community-level attributes. Numerous correlative studies have shown the density of many amphibian species to be lower in clearcuts compared to mature forests. Terrestrial salamanders, in particular, appear to be reduced by harvesting practices, with a median 3.5-fold difference in abundance between clearcut and forested sites (deMaynadier and Hunter 1995). However, the reasons for the observed lower densities are unknown.

One particular habitat component within forest ecosystems that may provide optimal habitat for terrestrial amphibians is the riparian zone. The riparian zone, defined as the interface between terrestrial and aquatic ecosystems, has a unique set of environmental conditions that may favour amphibian species, unlike those conditions found in upslope areas (Brosofske et al. 1997). A few studies have shown that riparian areas may be very important to amphibian populations (e.g., McComb et al. 1993, Gomez and Anthony 1996); however, it is unknown whether this is a broad pattern across regions and seasons, or if it is restricted to specific sites and/or environmental conditions.

Limited quantitative research has been conducted on British Columbia's terrestrial amphibian communities, and next to nothing in second-growth forest, the seral stage dominating southwestern British Columbia's landscape, where high densities of terrestrial amphibians are found. Therefore, the purpose of this study was to establish baseline conditions of abundance and movement patterns in riparian areas for terrestrial amphibian communities in 6 second-growth sites slated for harvesting. The specific objectives were: 1) to compare variation in abundance within and between 4 terrestrial salamanders among 6 second-growth sites; 2) to compare and contrast the movement patterns of aquatic-breeding salamanders with terrestrial-breeding salamanders; and 3) to determine if stream proximity affected the spatial distribution of terrestrial salamanders.

STUDY SITES

This study was conducted in the University of British Columbia's Malcolm Knapp Research Forest (122°34'W, 49°16'N), located approximately 60 km east of Vancouver.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1. General description of 6 second-growth sites in the Malcolm Knapp Research Forest.

Site	CWD[a]	Understory	Proximity of pond
Mike	large	present	<500 m
Spring	small	present	>1 km
H	medium	present	>1 km
Sck	medium	absent	>1 km
B	small	present	>1 km
I	large	absent	<500 m

[a] CWD = amounts of coarse woody debris at each site relative to each other.

This location is within the Coastal Western Hemlock biogeoclimatic zone, characterized by rainy, cool winters and dry summers. Six sites were selected in naturally regenerated 70-year-old second-growth forests, all of which were logged in the early part of the century and then burned in a stand-replacing fire. General characteristics of each site are described in Table 1. Each site will receive one of 3 treatments: control, 30-m buffer, and clearcut. The data presented in this paper include pretreatment results only.

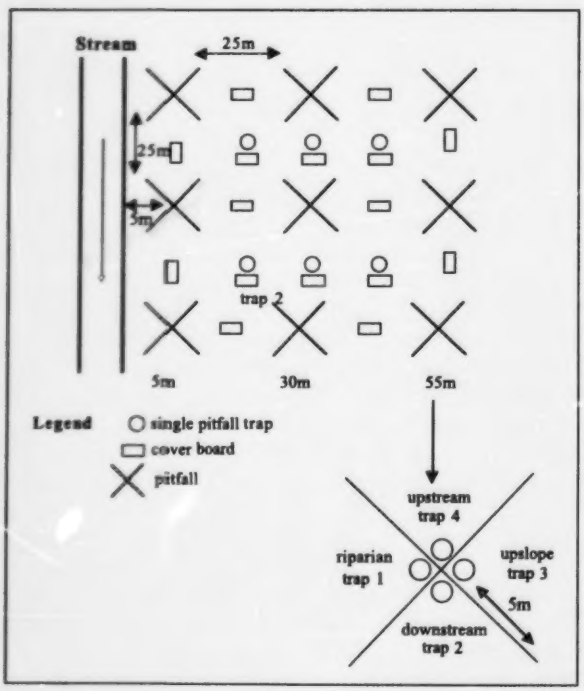

Figure 1. Arrangement of trapping grids designed to determine terrestrial amphibian abundance, direction of movement, and spatial distribution relative to the stream. Enlargement at bottom right illustrates design of a single array and arrangement of pitfall traps.

METHODS

Trapping and Marking

A combination of pitfall arrays, single pitfall traps, and cover boards were set up in a grid encompassing an area of approximately 50 X 50 m at each site (Fig. 1). Nine arrays consisting of 4 pitfall traps and drift fences were installed such that 1 trap each was oriented both downslope (trap 1) and upslope (trap 3) of the stream, as well as in the downstream (trap 2) and upstream (trap 4) direction (Fig. 1). The traps were intersected by a total of 20 m of drift fencing arranged in a cross, with the length of each arm equal to approximately 5 m. This X-arrangement of the pitfall arrays was designed to determine the direction of movement of the amphibians captured relative to the stream. Sixteen cover boards and 6 single pitfall traps were installed in the spaces between the arrays to increase amphibian capture probabilities.

Trapping was initiated in September 1997, with traps open for a total of 3 days at each site. In October and November, traps were opened for 2 or 3 days, alternating every 2 weeks and totalling 5 days of monitoring per month at each site. Trapping recommenced in February 1998, with grids monitored for 3 days, alternating every 2 weeks, for a total of 6 trap-nights per month. On 21 April 1998, traps were open continuously until 29 June 1998, when they were closed due to low trapping rate. Traps were opened a final time in July for a period of 4 trap-nights during a rain event.

Two marking techniques were used, depending on the species and weight of the individual captured. All salamanders and frogs with a weight of <8 g were marked in the field using a unique combination of elastomer dyes and toe clips. PIT (passively induced transponder) tags were used to individually mark 2 species of amphibians: northwestern salamander (*Ambystoma gracile*) and adult rough-skinned newt (*Taricha granulosa*). Animals to be PIT-tagged were taken back to the lab and anaesthetized using MS222; a small incision was then made in the lower abdomen, a PIT tag was inserted, and, finally, measurements were taken. The animals were kept overnight in a refrigerator to ensure their full recovery and then returned to their point of capture the following day. The following information was recorded for each amphibian captured: species, date, location of capture, snout–vent length, total length, mass, sex (if possible), age (juvenile or adult), individual mark, and whether the animal was newly caught or a recapture. Following processing, each individual was released approximately 1 m from the array on the opposite side from where it was captured, assuming the animal was moving in that direction.

RESULTS

Species Composition and Relative Abundance

A total of 1,247 amphibians were captured, including 5 salamander species and 3 anurans (Table 2); chorus frogs were

Table 2. Summary of total number of amphibians captured at all 6 sites combined.

Species	Common name	No. captured	No. of sites where captured
SALAMANDERS		979	
Ambystoma gracile	Northwestern salamander	327	6
Ambystoma macrodactylum	Long-toed salamander	15	3
Ensatina eschscholtzii	Ensatina	265	6
Plethodon vehiculum	Western redback salamander	232	6
Taricha granulosa	Rough-skinned newt	140	6
ANURANS		268	
Ascaphus truei	Tailed-frog	67	5
Bufo boreas	Western toad	167	6
Rana aurora	Red-legged frog	34	5
TOTAL		1,247	

present on at least 1 site but not captured. Although species composition was similar among sites, the relative abundance of each species was variable between sites and seasons (Figs. 2a–d). The aquatic breeders, northwestern salamander and rough-skinned newt, had similar patterns of abundance and seasonal activity, but differed from terrestrial-breeding salamanders in both respects. Both species were most abundant at Site I, with Mike Creek also having high numbers of northwestern salamanders. The timing of the activity patterns of the 2 aquatic breeders was slightly staggered. The northwestern salamanders' peak of activity occurred in the winter and spring, corresponding with their seasonal breeding migration. The rough-skinned newts' capture rate peaked slightly later than that of the northwestern salamanders in the spring, with movement frequency declining only slightly with the onset of summer. These salamanders had high coefficients of variation in capture rates between sites by season, ranging from 76.1% to 188.4% for the northwestern salamander, and from 64.5% to 162.2% for the rough-skinned newt.

Terrestrial-breeding salamanders, ensatina (*Ensatina eschscholtzii*) and western redback salamander (*Plethodon vehiculum*), responded similarly to site differences as reflected by their relative abundance at each site, as well as having similar patterns of seasonal activity. The abundance of terrestrial-breeding salamanders was highest at the same sites (Sck, B, and H) during the fall, contrary to the aquatic salamanders, which were most active in the winter and spring. Both the ensatina and western redback salamander were almost completely inactive in the winter, with another peak of activity occurring in the spring. There was a general decline of activity with the onset of the summer months at most sites for these 2 species. The coefficient of variation of capture rates between sites by season range from 57.5% to 245% for the ensatina, and from 44.2% to 154.9% for the western redback, reflecting the high site-to-site variation. The winter has the most variation in capture rates for the northwestern salamander, ensatina, and western redback salamander.

Movement and Spatial Distribution Relative to the Stream

For each species, the direction of movement was analyzed relative to the stream. The aquatic-breeding salamanders moved significantly more in the upstream/downstream direction of the stream: northwestern salamander: $\chi^2 = 16.12$, $P < 0.001$; rough-skinned newt: $\chi^2 = 14.33$, $P < 0.002$ (Fig. 3). No significant difference was detected in the movement direction for either the ensatina or western redback salamander.

The minimum average distance moved was calculated for each species based on recapture data (Fig. 4). The mean minimum distance moved by northwestern salamanders and rough-skinned newts was 32.1 m and 43.0 m, respectively. Ensatinas moved an average of 6.4 m and western redbacks an average of 0.64 m. Aquatic breeders moved significantly greater distances than terrestrial-breeding salamanders (1-way ANOVA, $F_{1,87} = 80.64$, $P < 0.001$).

The number of amphibians captured at 3 distances (5 m, 30 m, and 55 m) from the stream was analyzed both within and between seasons to determine if stream proximity affected the salamanders' spatial distribution (Figs. 5a–d). Within seasons, the northwestern salamander was captured significantly more often at 30 m from the stream during both the fall ($\chi^2 = 7.52$, $P < 0.02$) and winter ($\chi^2 = 9.08$, $P < 0.01$) than at either 5 m or 55 m. There was no significant difference in the number captured between the 3 distances for any other species. Between seasons, there was a trend toward increased capture rates near the stream during the summer for all species, but this was not significant.

DISCUSSION

Numerous studies conducted in the Pacific Northwest have found amphibian abundance to be highly variable within and among stand ages ranging from clearcut to old-growth forests (e.g., Aubry and Hall 1991, Corn and Bury 1991). Experimental results from studies using this chronosequential

approach are confounded because variation in amphibian abundance across stand ages due to site-specific differences cannot be distinguished from variation induced by treatment effects. Conclusions may be further limited by the lack of information on the movement patterns of amphibian species both within "natural" systems and in human-altered landscapes (Gibbs 1998). There is some debate as to whether amphibians emigrate from the disturbed sites or migrate vertically into the substrate, aestivating and eventually dying due to the altered environmental conditions. While death and emigration are 2 mechanisms explaining the pattern of lower abundance in clearcut sites versus forested areas, management strategies for terrestrial amphibians might be quite different depending on the process actually responsible for the patterns observed. Therefore, the collection of pretreatment data, including information on life history parameters and movement patterns, are critical in determining the response of terrestrial amphibians to forest harvesting.

The variation in capture rates, movement patterns, and minimum distance moved for aquatic-breeding salamanders is expected based on their life history. The spatial distribution of northwestern salamanders and rough-skinned newts is dependent upon the proximity of suitable breeding sites. Site I and Mike Creek have large ponds within a few hundred meters of the study sites; the high number of aquatic breeders captured at these sites reflect the large population of salamanders these ponds are capable of producing and supporting. High variation in capture rates between forested sites of similar ages were also observed by Aubry and Hall (1991) and Corn and Bury (1991), and postulated to be a function of distance from suitable breeding sites and not a response to forest age per se. In addition, both species conduct annual breeding migrations in the late winter, spring, and early summer in which direction of movement is nonrandom, but oriented towards or away from breeding ponds (Blaustein et al. 1995). The pattern of movement of both salamanders parallel to the stream in this study is a reflection of these migrations. Both species are capable of moving several hundred meters from their terrestrial refuges to breeding ponds and, therefore, can easily move through the

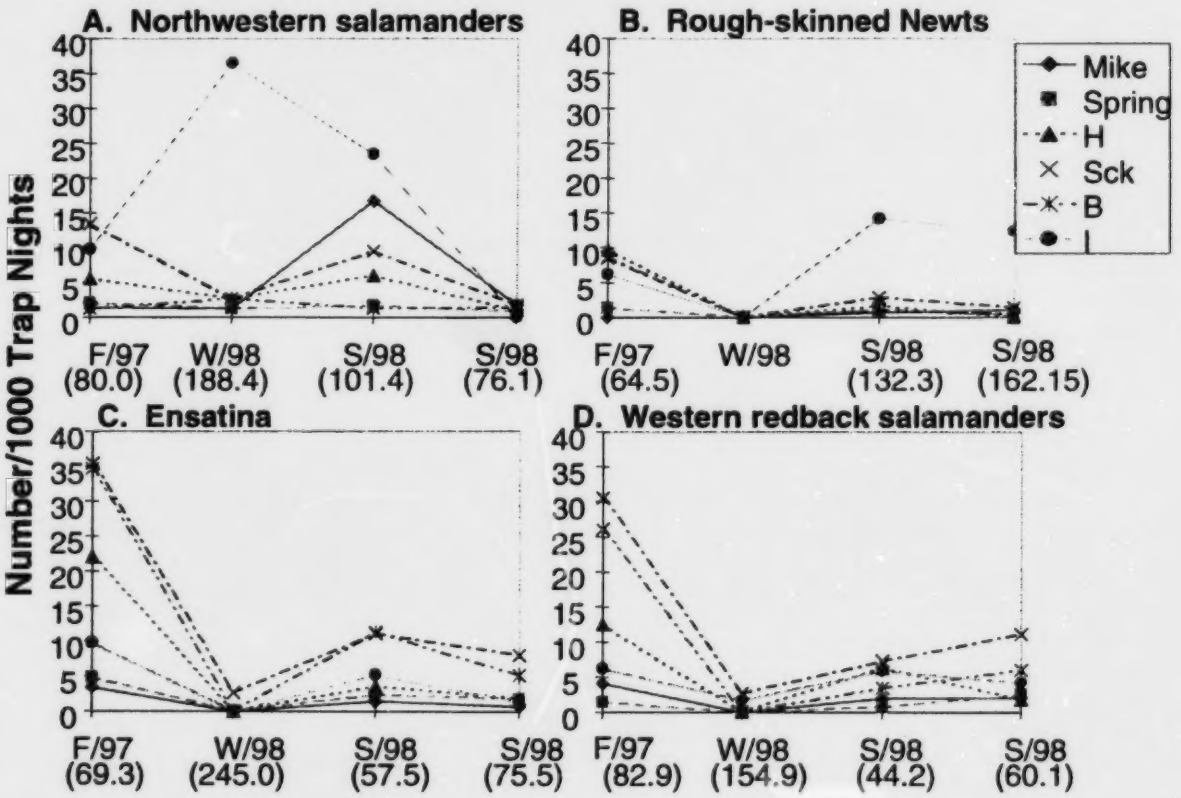

Figure 2a–d. Relative abundance of 4 terrestrial salamanders at 6 sites by season. Coefficient of variation between sites by season is included in brackets.

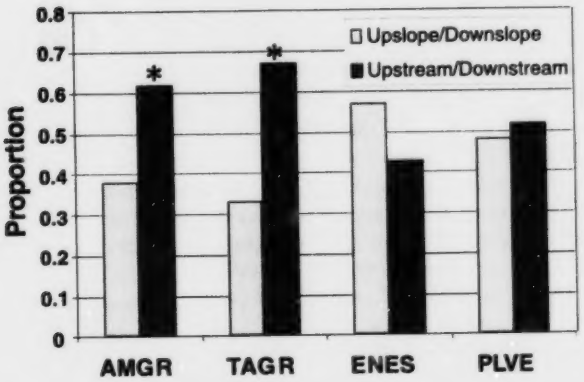

Figure 3. Comparison of the parallel versus perpendicular movement of 4 terrestrial salamanders. (Aquatic breeders: AMGR - northwestern salamander; χ^2 = 16.12 , $P < 0.001$; TAGR - rough-skinned newt, χ^2 = 14.33 , $P < 0.002$. Terrestrial breeders: ENES - ensatina, $\chi^2 = 7.09$, $P > 0.05$; PLVE - western redback salamander, $\chi^2 = 0.14$, $P > 0.05$.).

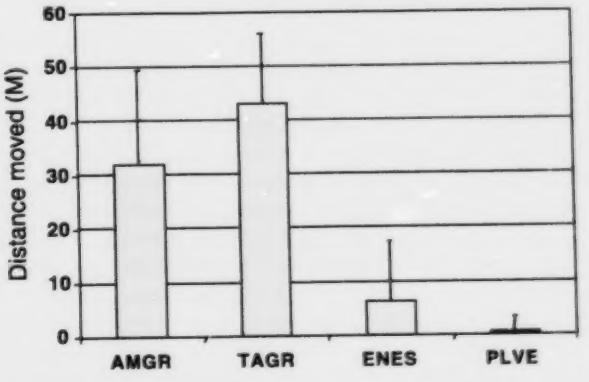

Figure 4. The minimum average distance moved for the 4 terrestrial salamanders. (Aquatic breeders: AMGR - northwestern salamander, and TAGR - rough-skinned newt. Terrestrial breeders: ENES - ensatina, and PLVE - western redback salamander).

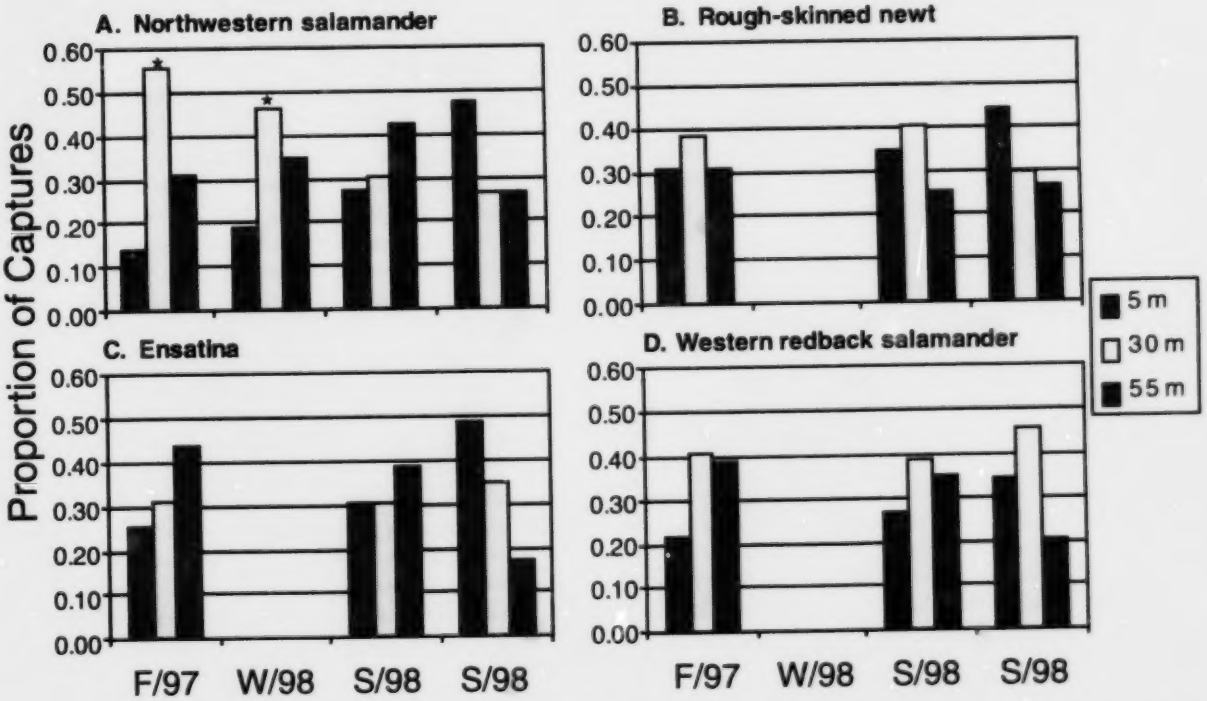

Figure 5a–d. Proportion of captures with increasing distance from the stream. Within seasons there was a significant difference in the proportion captured at the 3 distances from the stream for northwestern salamander in Fall 1997 ($\chi^2 = 7.52$, $P < 0.02$) and Winter 1998 ($\chi^2 = 9.08$, $P < 0.01$). No other species showed a significant pattern of abundance relative to distance from the stream between seasons (χ^2 test, all with $P > 0.05$).

50-m grids used in this study, as indicated by the high minimum average distance recorded for each species.

The apparent random movement of terrestrial-breeding salamanders, ensatina and western redback, and the low average distances moved are also expected based on their life histories; the high site-to-site variation in abundance is more difficult to explain. Both the ensatina and western redback salamander are members of the family Plethodontidae, which is characterized by its completely terrestrial existence. Both species breed terrestrially, laying their eggs in burrows or decayed wood, or on talus slopes, in the spring. Development occurs through the summer with hatchlings emerging in the fall as miniature adults (Blaustein et al. 1995). Moist substrate conditions are required for successful reproduction, but standing water is not necessary for either salamander and, therefore, they make no annual breeding migrations. Habitat requirements for each species may be met in a much smaller area than for the aquatic breeders. The observed mean minimum distance moved of 0.64 m for western redback salamanders is similar to the distances observed by Ovaska (1988) who calculated the home range size of western redback salamanders to be approximately 2.5 m^2, with 8.5 m being the maximum recorded distance moved by 1 individual. The mean distance of 6.4 m recorded for ensatinas is less than that observed by Staub et al. (1995), who calculated mean movement distances of 21.7 m and 22.2 m for males and females, respectively. This discrepancy in distance is likely an artifact of the different sampling designs used for both studies.

As there were no obvious differences between the 6 study sites, and because terrestrial-breeding salamanders require only 1 habitat to successfully survive and reproduce, the observed variation in capture rates between sites was not expected. However, there are 2 hypotheses potentially explaining the observed variation. One hypothesis is that competition and/or predation by the larger, aquatic-breeding salamanders may limit the abundance of the terrestrial breeders on at least 2 of the study sites. Both the ensatina and western redback salamander are most abundant on sites where northwestern salamanders and rough-skinned newts are least abundant. Competition and predation have both been experimentally determined to affect the distribution of terrestrial salamanders in eastern North American amphibian communities (Hairston 1996) and, therefore, cannot be ruled out as mechanisms structuring amphibian communities on these study sites. A second hypothesis explaining the site-to-site variation may be inherent differences in site quality, such as soil moisture or productivity, which limit the density, reproductive rates, and/or survival rates of salamanders. For example, eastern redback salamanders (*Plethodon cinereus*) can sense low soil pH and move away from areas where pH is too low (Wyman and Hawksley-Lescault 1987). Salamanders could be responding to such environmental factors, limiting their abundance on certain sites in this study.

Importance of Riparian Areas

Due to the physiological requirements of amphibian species, riparian areas have been suggested to provide optimal habitat for amphibians (deMaynadier and Hunter 1995). Humidity and temperature are 2 important factors determining the distribution of many amphibian species; higher humidity and lower temperatures are often associated with stream presence (Brosofske et al. 1997). In this study, however, stream proximity did not appear to affect the spatial distribution for any of the salamanders. These results are contrary to those of McComb et al. (1993) and Gomez and Anthony (1996), who both observed higher capture rates along streams versus upslope areas within Pacific Northwest coniferous forests. This discrepancy in results may reflect a difference in the size of the streams used in the various studies. Larger streams have greater influence on the surrounding terrestrial environment than do smaller streams. McComb et al. (1993) and Gomez and Anthony (1996) both worked on second- to fourth-order streams averaging >2 m in wetted width. These streams are larger than those used in this study and, therefore, amphibians may not have had a strong trans-riparian gradient to respond to. Secondly, the scales of the studies are much different. Riparian captures were within 10 m of the stream and upslope captures were 200 m from the stream for Gomez and Anthony (1996), and 400 m for McComb et al (1993). Upslope in this study is only 55 m from the stream, which may also explain the discrepancy.

Although terrestrial salamanders are not considered obligate riparian users, they still may benefit from the retention of buffer strips along headwater streams. As indicated by previous comparative studies (e.g., Aubry and Hall 1991, Corn and Bury 1991), all 4 salamander species found on our study sites may respond negatively to forest harvesting. The non-random movements of both northwestern salamanders and rough-skinned newts, parallel to the stream, indicate buffer strips may potentially be used as corridors, although whether they will be preferentially selected over clearcut areas is unknown, but hopefully will be elucidated with our posttreatment data. In the case of terrestrial-breeding amphibians, buffer strips may act as refuge areas for the population. Leave strips may allow the population to persist until the adjacent clearcut has recovered to provide suitable habitat. A nearby population source may be especially important due to their limited movement capabilities and high site fidelity, particularly for western redback salamanders. Plethodontids have evolved under the stable, moderate conditions that old-growth forests provide. Density-independent perturbations, such as clearcutting, make it difficult for salamander populations with low fecundity and vagility to recover following the disturbance.

ACKNOWLEDGEMENTS

This research was funded by Forest Renewal British Columbia and the Habitat Conservation Trust Fund. K. Claire, R. Scott, D. O'Donahue, and numerous other volunteers are gratefully acknowledged for their assistance in the field.

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Terrestrial Pacific Giant Salamanders: Natural History and Response to Forest Practices

Barbara E. Johnston
Department of Zoology, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada
johnston@fisheries.com

ABSTRACT

The Pacific giant salamander (*Dicamptodon tenebrosus*) is Red-listed in British Columbia because of its limited geographic distribution and loss of its habitat through logging and urban development. Aspects of larval giant salamander ecology have been studied, but virtually nothing is known about the terrestrial phase of this species. I used radiotelemetry to examine movements and habitat use of terrestrial Pacific giant salamanders in forested habitat. I also investigated the effects of clearcut logging and riparian buffer strips on their behaviour. Terrestrial Pacific giant salamanders were found to be relatively sedentary creatures that spent most of their time in refuges such as burrows, rotten logs, and streams. They were predominantly nocturnal and their activity level was strongly associated with rain. Although their response to forest practices was ambiguous, some results suggested that these animals may be adversely affected by clearcut logging and that riparian buffer strips could help conservation efforts.

Key words: *Dicamptodon tenebrosus*, habitat, logging, Pacific giant salamander, radiotelemetry.

The Pacific giant salamander is generally considered to be an obligate associate of old-growth forests, most commonly found along fast-flowing mountain streams. This semiaquatic species depends on streams for breeding and larval development. Although the ecology of aquatic Pacific giant salamanders has been studied, little is known about the terrestrial phase of this species.

In Canada, the Pacific giant salamander is found solely within the Chilliwack River valley. Because of its limited geographic distribution and the rapid rate at which its habitat is being lost to logging and residential development, the Pacific giant salamander is Red-listed (considered threatened or endangered) in British Columbia and is classified as "vulnerable" by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). To better manage for the conservation of the Pacific giant salamander in British Columbia we must learn more about its basic natural history and the effects that habitat alteration have on this elusive creature. My objectives were to: (1) increase our understanding of terrestrial Pacific giant salamander natural history, specifically habitat use and activity pattern; (2) determine the effects of clearcut logging on movement patterns and habitat use of terrestrial giant salamanders; and (3) assess the efficacy of riparian buffer strips for their conservation.

METHODS

The study was conducted during the summer and fall of 1996 and 1997 in the Chilliwack and neighbouring Nooksack river valleys in southwestern British Columbia and northwestern Washington. I used 7 study sites: 2 old growth, 2 second growth, 1 clearcut to the stream margin, and 2 clearcut with forested riparian buffer strips (20–30 m wide) retained on each side of the stream.

Twenty salamanders with mass >25 g were captured during night searches and each was surgically implanted with a radio transmitter (Holohil model BD-2GII;Holohil Systems Ltd., Carp, Ont.). Salamanders were then released at their capture sites. Using a handheld directional antenna, I located each individual every second day for the remainder of the study (about 3 months). Telemetry locations were used to determine movement patterns and large-scale habitat use of the salamanders. I also conducted 24-hour monitoring sessions to determine daily activity patterns. I measured microhabitat features at 10 refuge sites per salamander and compared these data with information collected at paired "nonuse" sites used to represent habitat availability.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

RESULTS

Natural History

Movements

The terrestrial Pacific giant salamanders were relatively sedentary creatures. They often went several days or even weeks without changing location. They were predominantly nocturnal and their activity level was strongly associated with precipitation. During periods without rain, activity was negatively associated with temperature. No evidence suggested seasonal migration in this species, as is characteristic of many closely related ambystomid salamanders.

Habitat Use

There was a great deal of individual variation in the way salamanders used their habitat. Some salamanders never ventured away from the stream while others were found primarily in upslope habitat. Refuge sites were associated with the availability of coarse woody debris, water, rocks, and leaf litter. Salamanders were also frequently found underground in small mammal burrows and root channels. When using coarse woody debris for cover, salamanders selected old wood in advanced stages of decay over newly fallen wood.

Effects of Clearcut Logging

Because these animals are fossorial and relatively sedentary, they are difficult to capture using the standard amphibian techniques (pitfall traps, time and area constrained searches). Therefore, I could not compare population densities or relative abundance in logged and control areas. I used 2 alternate approaches to assess the effects of clearcut logging on terrestrial Pacific giant salamanders. First, I examined the response of salamanders when they encountered the forest-clearcut interface. At sites with riparian buffer strips I released radio-tagged salamanders right at the boundary between forest and clearcut habitat. These salamanders did not move in random directions, but avoided entering clearcut habitat.

Second, I compared the behaviour of giant salamanders in different habitat types. For this analysis I measured total area covered, mean movement length, mean refuge duration, seasonal activity level, diel activity pattern, mean distance from stream, and proportional use of near-stream versus upslope habitat. While some measures showed no effect of habitat type, trends in other measures suggested that salamanders in clearcut habitat might have adjusted their behaviour to reduce their risk of desiccation. For example, compared with salamanders in forested areas, individuals in clearcut habitat covered less area, stayed in refuges longer, reduced their activity level in the dry summer season, and stayed closer to the stream. Salamander activity was also more dependent on precipitation in clearcuts than in other habitat types.

Efficacy of Riparian Buffer Strips

Riparian buffer strips appeared to be a promising management strategy for the conservation of terrestrial Pacific giant salamanders. A hypothetical buffer 40 m on each side of the stream contained 88% of the telemetry locations recorded for salamanders at forested sites. A buffer half this width (20 m on each side of the stream) still contained 80% of salamander locations. In addition, the behaviour of salamanders at sites with riparian buffer strips was indistinguishable from that of salamanders found in continuous forest.

DISCUSSION

Although terrestrial Pacific giant salamanders occur in clearcuts, individuals in this habitat appear to alter their behaviour in ways expected of amphibians under moisture or temperature stress. Without information about giant salamander survival or reproductive rates, effects of clearcut logging on salamander fitness and long-term population persistence remain to be determined.

ACKNOWLEDGEMENTS

This research was funded by Forest Renewal BC, the BC Habitat Conservation Trust Fund, and World Wildlife Fund Canada. I thank my supervisor W. Neill and my field assistants L. Frid, M. Krause, and D. Casimir.

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Bat Usage of the Weymer Creek Cave Systems on Northern Vancouver Island

Martin J. Davis
Island Karst Research
Box 8124, Stn. Central PO, Victoria, BC, V8W 3R8, Canada
iskar@pacificcoast.net

Alisa D. Vanderberg
Island Karst Research
Box 8124, Stn. Central PO
Victoria, BC, V8W 3R8, Canada

Trudy A. Chatwin
British Columbia Ministry of
Environment, Lands and Parks
2080 Labieux Road
Nanaimo, BC, V9T 6J9, Canada

Monica H. Mather[1]
3165 Rock City Road
Nanaimo, BC, V9T 1T5, Canada

ABSTRACT

Bat use of the Weymer Creek karst area, Vancouver Island, British Columbia, Canada, was studied from 1996 to 1998, following the discovery of a *Myotis* bat hibernaculum. We used a variety of monitoring techniques backed up by careful observations to develop an understanding of how bats use the study area and to determine the climatic characteristics of subsurface bat habitat. We hope to produce recommendations that will lead to sound management of *Myotis* bats and their habitat within the west coast rain forest karst environment. The Weymer caves are used for swarming and hibernation by *Myotis volans*, *M. lucifugus*, *M. yumanensis*, and the rare and endangered *M. keenii*. In addition, *M. californicus* and *Eptesicus fuscus* use the surface habitat for feeding. Caves used for swarming and hibernation vary in elevation from 500 to 900 m, with the largest concentrations of hibernating bats being found above 800 m. Most hibernating bats are found in deep passages with stable environments of 3–4°C and 100% relative humidity. Caves underlying clearcuts are also used by bats and have stable winter temperatures. Within the study area, female Keen's long-eared myotis use low elevation sites (<400 m) in summer for feeding and raising young. They join males in cave swarming activity at higher elevations (>650 m) between August and September. Our observations suggest that the entire range of habitats from sea level to high elevation and both cave and surface habitats are vital to the life history and survival of at least 6 species of bats.

Key words: bats, caves, hibernation, karst, *Myotis keenii*, Vancouver Island.

Nine species of *Myotis* bats inhabit British Columbia in summer, yet until 1993, no winter aggregation sites were known for any *Myotis* in British Columbia (Nagorsen and Brigham 1993). Any new information regarding critical hibernaculum sites and characteristics of these sites is vital to bat management and conservation. The Province of British Columbia is initiating legislation and guidelines to regulate forest practices and ensure protection of forest-dependent endangered species at this time; however, due to the lack of information about forest-using bats, recommendations are based on information from other jurisdictions. Thus, in 1993, when cavers discovered hibernating *M. lucifugus*, *M. volans*, and the endangered *M. keenii* in Labyrinth Cave under a Vancouver Island coastal montane forest slated for logging, measures were taken to protect and study the Weymer Creek cave systems.

Because the Labyrinth hibernaculum had its main entrance at 900 m above sea level, we hypothesized that bat hibernation in coastal areas may be related to the constant cool temperature in the higher elevation caves during hibernation. If this proved true, then efforts to locate and conserve bat hibernacula in coastal karst areas could focus on higher elevation sites. Our goal was to determine critical habitat characteristics for bats in this coastal rain forest ecosystem to make recommendations for conserving bats and bat habitat within this environment. Our specific objectives are

1. to document cave usage by bats in the Weymer karst area through discreet visitation, electronic monitoring, guano collection, and skeleton collection;

[1] Present address: 7017 Lavender Road, Lantzville, BC, V0R 2H0, Canada

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

2. to determine the physical characteristics of bat hibernacula through year-round monitoring of temperature and humidity patterns in caves of different elevation ranges;
3. to document summer habitat use patterns of bats in caves and the overlying forest; and
4. to determine the impact of clearcut logging on cave physical characteristics and bat use patterns.

STUDY AREA

The Weymer and Green Creek drainages, located about 3 km southeast of Tahsis on northwestern Vancouver Island, range from sea level to 1126 m (49°51'N–49°53'N, 126°37'W–126°39.5'W). The study area contains >150 caves from sea level to 1,000 m, and was partially clearcut over 25 years ago (Davis 1995). The study area is primarily in the Coastal Western Hemlock (CWH) biogeoclimatic zone (subzones vm1 and vm2) and Mountain Hemlock (MH) biogeoclimatic zone (subzone mm1). Lower elevations have small stands of old-growth Douglas-fir (*Pseudotsuga menziesii*), while red alder (*Alnus rubra*) and bigleaf maple (*Acer macrophyllum*) predominate in areas that were logged before 1950. The cave systems are mostly protected within a 305-ha park, while the remainder are within Tree Farm Licence (TFL) 19.

METHODS

Temperature and Humidity Data Logging

To characterize the physical habitats suitable for bat hibernation we deployed "Onset" temperature and humidity data loggers in caves and surface control sites at low elevation (0–400 m), mid-elevation (401–650 m), and high elevation (651–950 m). For low elevation, finding replicate caves of sufficient length was not always possible, so some shorter caves were used. Data were also collected from caves under clearcuts and under forested areas in each elevation category. The pair of temperature and humidity loggers were

Table 1. Summary of mean winter temperatures and coefficients of variation (CV) at data logger sites (sorted by mean temperature). Data from 17 October 1996 to 5 March 1997.

Cave site	Mean temp (°C)	CV	Cave depth (m)	Elev. cat.[a]	Elevation (m)	Treatment[b]	Hibern. bats
Wormhole: entrance	0.05	60	10	H	810	cc	
Labyrinth: snag	0.23	4.64	0	H	922	fo	
Labyrinth: entrance	0.52	3.55	10	H	915	fo	
Marmot Maus: snag	1	3.94	0	H	824	cc	
Slot Canyon: snag	1.43	1.951	0	M	596	fo	
Whistling: clearcut	1.65	2.273	0	M	568	cc	
Whistling: low entrance	1.83	1.361	10	M	510	cc	
Headwall: entrance	2.39	0.842	10	M	483	fo	
Labyrinth-Keens Junct.	2.64	0.114	100	H	843	fo	yes
Wormhole: station 19	2.68	0.198	100	H	751	cc	yes
Slot Canyon: Mcl. ent.	2.77	0.339	10	M	554	fo	
Labyrinth: hibernaculum	3.08	0.026	200	H	838	fo	yes
Slot Canyon-Rckn. Rub.	3.56	0.242	300	M	592	fo	yes
Deerdrop-Ant Lion	3.98	0.02	150	H	788	fo	yes
Whistling: junction	4.17	0.218	100	M	531	cc	
Deerdrop: entrance	4.25	0.136	10	H	820	fo	
Boneyard: outside	4.26	0.6	0	L	5	fo	
Headwall-Redrock	4.67	0.112	100	M	490	fo	
Marmot Maus.: pit	4.68	0.034	25	H	798	cc	
Marmot Maus.: entrance	4.79	0.087	10	H	813	cc	
Crab: 20 m in	4.87	0.335	20	L	60	cc	
Whistling: Canal/sump	5	0.059	100	M	565	cc	
Whistling: mid-entrance	5.35	0.258	10	M	564	cc	
Cave176a: 15 m	5.52	0.287	15	L	50	fo	
Knollhole: 10 m in	6.59	0.264	10	L	65	cc	
Knollhole: 30 m in	7.8	0.043	30	L	50	cc	
Boneyard: tidal sump	7.97	0.053	10	L	3	fo	
Boneyard: mouse dig	8.13	0.019	50	L	6	fo	

[a] Elevation categories: 0–400 m above sea level = LOW (L); 401–650 m = MID (M); 651–950 m = HIGH (H).
[b] Treatment categories: fo = forested, cc = clearcut.

placed 10 m within the cave entrance and deep within the cave (where possible >100 m from the entrance). Surface temperatures were monitored in each treatment category. Data were recorded at 1-hour intervals.

We compared temperatures from data loggers in 28 locations (Table 1). To compare temperature differences and variations throughout the winter, we used data collected between 17 October 1996 and 5 March 1997. Analysis of temperature data included calculating seasonal means and standard errors at logger locations. To compare temperature variability, we calculated the coefficient of variation (CV), a dimensionless quantity that measures the amount of variability relative to the value of the mean; the CV is calculated by dividing the standard deviation by the mean. We used analysis of variance (ANOVA) and Student's *t*-test (1) to compare mean temperatures and CVs among caves at different elevations and depths to determine if temperatures were constant and within ranges suitable for hibernating bats, and (2) to compare temperatures and variability in caves under clearcuts and those in forested areas. We compared mean temperatures and variability with elevation using simple regression analysis.

Bat Activity

We delineated the sampling periods based on seasonal patterns of bat use and the net-nights when we first and last noted them: hibernation (mid-October to late May), emergence from hibernation and feeding in early summer (20 May to 8 July), and swarming in late summer (29 July to 13 September).

To document surface and subsurface habitat use by bats we used both direct and indirect inspection methods: hibernation sites were confirmed by cave visits and exploration by cavers; skeletal remains and recently dead bats were collected from caves to confirm bat use and identify species; guano collection mats were deployed in caves to confirm current bat use; remote ultrasonic detectors were deployed to monitor and record bat passes at entrances and selected surface sites; and mist-netting was used at forest edges, ponds, and outside cave entrances to identify species and habitat use.

In 1998, we netted at various elevations and habitat types from 20 May to 10 October, on 82 net-nights. Bats were handled according to Resources Inventory Committee (RIC) standards (Resources Inventory Committee 1998)

Bone identification of bat species, especially *M. keenii*, was based on skull measurements and characteristics defined in van Zyll de Jong and Nagorsen (1994).

RESULTS

Winter Temperature Means

The average winter temperature at our 28 data logger sites ranged from 0 to 8°C (Table 1). Mean temperatures were significantly lower at high elevations (simple regression, df = 27, $F = 28.6$, $r^2 = 52\%$, $P < 0.05$, Fig. 1A). Temperatures were also significantly higher deep within caves (approx. 100 m) compared with those outside caves (ANOVA, df = 24, $F = 5.14$, $P < 0.05$; Fig. 2A). There was no significant difference between temperatures near caves in clearcuts versus those in forested areas (*t*-test, $t = -1.0$, df = 23, $P > 0.05$, Fig. 2A).

Winter Temperature Variation

The mean CVs were greater at higher elevations (simple regression, df = 26, $F = 4.25$, $r^2 = 15\%$, $P < 0.05$, Fig. 1B). The CV was also significantly different among cave depth categories. Locations outside caves and at entrances (0–10 m) had higher temperature variations than did deep (50–200 m) within caves (ANOVA, df = 24, $F = 3.44$, $P < 0.05$, Scheffe's pairwise comparisons, depth category 0–10 m differs from 15–45 m, and from 50–200 m, $P < 0.05$; Fig. 2B). There was no significant difference in coefficient of variation between caves in clearcuts and forested areas (*t*-test, $t = 0.06$, df = 23, $P > 0.05$, Fig. 2B).

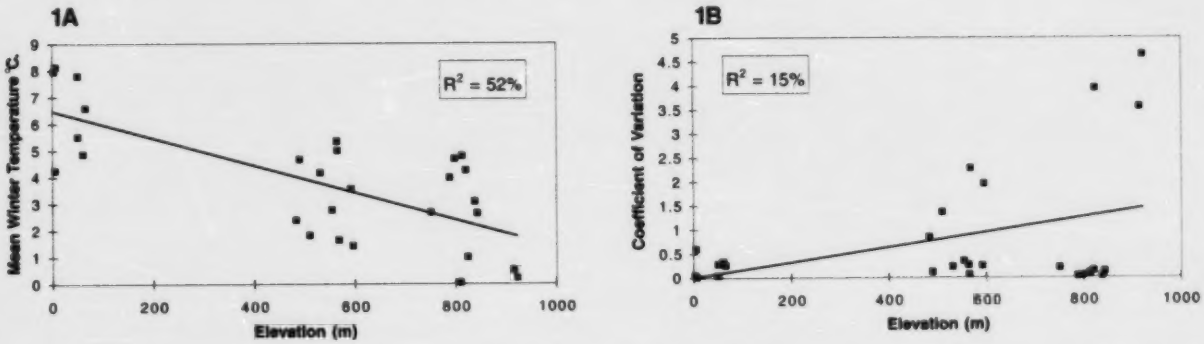

Figure 1. Simple regression relationships between (A) winter temperatures at cave sites and (B) coefficients of variation (CV) versus elevation ($p < 0.05$).

Table 2. Bat capture summary, 1998.

Category	Net-nights	Total captures[a]	No. females, juveniles	Capture ratio[b]
Caves: mid- to high, early summer	8	0		0
Surface: mid- to high, early summer	10	1, EPFU		0.1
Caves: low, early summer	1	0		0
Surface: low, early summer	3	1, MYKE	1, F	0.333
Caves: mid- to high, late summer	46	33, MYVO;15, MYLU, 9; MYKE; 2, MYYU, 4; MYLU/YU; 2, MYKE/EV TOTAL: 65	1, F 1, J(F) 3 J(M)	1.413
Surface: mid- to high, late summer	8	0		0
Caves: low, late summer	7	3, MYKE	2, F; 1, J(M)	0.429
Surface: low, late summer	4	0		0
TOTALS	87	70	5, F;5, J	0.805 (avg)

[a] EPFU, *Eptesicus fuscus*, big brown bat; MYKE, *Myotis keenii*, Keen's long-eared myotis; MYYU, *M. yumanensis*, Yuma myotis; MYLU, *M. lucifugus*, little brown myotis; MYLU/YU, bats that were indistinguishable between *M. lucifugus* and *M. yumanensis;* MYKE/EV, bats that could be *M. evotis*. A skull, collected from Weymer, and identified as a possible *M. evotis* (D. Nagorsen, pers. comm.) raises taxonomic questions.

[b] The number of bats caught divided by the number of net-nights (1 bat net/3 hour+ sampling period).

Hibernacula Temperatures and Humidity

We found evidence of bats hibernating in 5 deep cave locations between 592 and 843 m (Table 1). The average winter temperatures in the hibernacula ranged from 2.6 to 3.9°C. The average CVs ranged from 0.02 to 0.24 (Figs. 2A, 2B).

Because all hibernacula were found in deep cave sites we looked at the differences between mean temperature and variation in all deep cave sites at different elevations to determine consistent temperature characteristics of bat hibernacula. Deep cave sites at low elevations had much higher mean temperatures than did those at mid-elevation and those at high elevation (ANOVA, df = 11, $F = 28$, $P < 0.05$, Fig. 3A). The CV in deep caves did not vary with elevation (ANOVA, df = 10, $F = 2.16$, $P > 0.05$). The mean temperatures within deep caves with hibernating bats were significantly lower than those without (t-test, df = 9, $t = -3.1$, $P < 0.05$, Fig. 3B); however, the CV was not different for deep sites with and without hibernating bats (t-test, df = 9, $t = 0.75$, $P > 0.05$, Fig. 3B). We found no differences in mean temperatures or CVs between deep cave sites under clearcuts and under forests (mean: t-test, df = 9, $t = -0.45$, $P > 0.05$; CV: t-test, df = 9, $t = -0.61$, $P > 0.05$, Fig. 3B).

Relative humidity at the largest hibernacula (Labyrinth) was a constant 100% in winter. Other sites have not been analyzed yet.

Cave Habitat Use by Bats

During the summer/early fall season in 1996 and 1997 we captured 91 bats of 5 species over 21 net-nights: *Myotis volans*, 46; *M. keenii/evotis*, 6; *M. lucifugus*, 29; *M. lucifugus/yumanensis*,

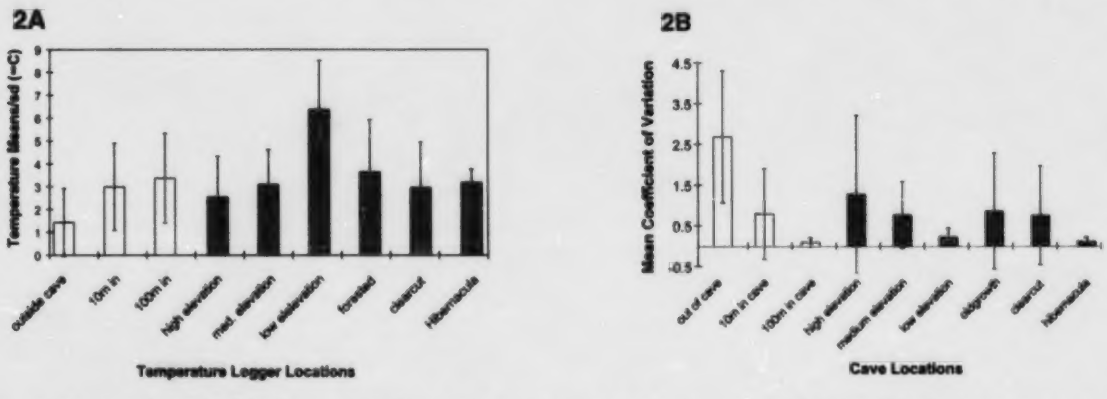

Figure 2. Means and standard deviations (SD) show variations in (A) winter temperature and (B) coefficient of variation in and near caves at different elevations, depths, and cover.

Table 3. Bat use of Weymer Caves.

Cave	Elev.[a] (m)	Length[b] (m)	Treatment[c]	Visual insp.	Ultrasonic detection[d]	Guano	Swarming[e]	Skeletal remains	Hibernation[f]
Cathedral	945	6,230+	fo	X[g]	X	X	X		X
Labyrinth	925	6,230+	fo	X	X	X	X	X	X
Keyhole	875	6,230+	fo	X		X	X	X	X
Calypso	800	6,230+	fo				X	X	X
Ursa Major	910	6,230+	fo		X			X	X
Waymore	840	119	fo				X		
471I	890	?	fo				X		X
Drafty Drop	885	75+	fo		X				
471	860	91	fo				X		
Fracture	840	163+	fo	X	X		X		X
Deer Drop	820		fo	X	X	X	X	X	X
Wormhole	760	1,583+	cc	X		X	X	X	X
Marmot Mausoleum	780	27	cc					X	
X Cave	705	97+	cc				X		X
Skypot	690	5,365+	cc		X		X		X
Boggle	740	358+	fo		X		X		X
Slot Canyon	550	5,365+	fo	X	X	X	X		X
Fallen Giant	540	1,680+	fo		X		X		
Headwall	480	1,680+	fo		X				
Whistling	560	782	cc		X				
Liquid Sky	380	385+	fo					X	
Knoll Hole	70	90	cc			X	X		
Crab Crevice	70	25	cc						
Cave 176A	60	17	fo						
Boneyard	4		fo						

[a] Elev. = entrance elevation. Elevation categories are: 0–400 m above sea level = LOW; 401–650 m = MID; 651–950 m = HIGH.
[b] "+" indicates cave is longer than mapped length.
[c] fo = forested; cc = clearcut.
[d] Indicates visual verification of hibernacula.
[e] Swarming indicates multiple bats flying in and out of caves from late July to mid-September.
[f] Hibernation indicates high probability of hibernacula. Hibernation period is mid-October to mid-June
[g] X indicates documented evidence of activity.

6; *M. yumanensis*, 3; and *M. californicus*, 1 (Davis 1997).

In 1998 we caught 70 bats of 5 species over 87 net-nights (Table 2). All but *Eptesicus fuscus* and *M. californicus* have been captured at cave entrances.

At mid- and high elevation caves in early summer, no bats were caught. Emergent bats were documented at high elevation cave entrances between 23 May and 27 May. ("Emergent" bats are bats that are found roosting or in torpor just inside the Weymer cave entrances in May–June, typically 10–30 m within. Individuals have been observed to remain for up to 2 weeks at the same location). However, we were surprised to find 2 bats, probably *M. volans*, hibernating in Wormhole Cave on 21 June, over 300 m from the nearest known entrances. This date was later than has previously been observed in the study area.

There was no recorded activity at low elevation caves in early summer during a single night's netting effort.

By late July, there was swarming activity (multiple bats flying in and out of caves that are used for hibernation) by male *Myotis* of 4 species (*M. volans*, *M. keenii*, *M. lucifugus*, *M. yumanensis*) at the mid- to high elevation caves (capture ratio: 1.4 bats/net-night) (Table 3). The greatest swarming activity occurred at entrances to the longer caves with the most stable temperatures, such as the Weymer System, Ursa Major System, and Wormhole Cave. Guano collection sheets record the highest rates of deposition in caves at high elevation (229 droppings on 18 sheets). At mid-elevation caves, we documented most bat activity at entrances that are lower exits to lengthy, deep cave systems that originate at higher elevations and have stable, cool airflows, such as the Slot Canyon entrances to the Weymer System. Caves with all entrances at mid-elevation, such as Fallen Giant, Headwall, and Whistling, yielded few bat captures and detections: 1 capture out of 5 net-nights and 18 passes out of 8 detector nights over 2 years. Guano collection sheets also showed little deposition at this elevation (15 droppings on 14 sheets).

Swarming activity in August began at 2330, whereas forest commuting and feeding activity began at dusk (about 2130). Swarming began progressively earlier as the day length shortened. By 2 September, bats were being caught by 2120. The swarming activity increased through August and by early September appeared to peak. Our highest capture ratio, 5.25 bats/net-night, occurred on 2 September. In 1996 and 1997, however, the swarming peaked in mid-August (11–14 Aug.). Females and juveniles did not appear with the swarming males until early September in 1998. The relative abundance of females for this period was only 3% that of males, over 3 years of netting. The overall capture ratio of bats at cave entrances during late summer was 1.28 bats/net-night.

The capture ratio for low elevation caves during late summer was 0.429. Two significant low elevation caves (Boneyard and Liquid Sky) exhibited no bat activity. However, a 100-m long cave, Knoll Hole, showed activity during netting on 11 August 1998 between 2030 and 2100. Two post-lactating females and 1 juvenile male *M. keenii* were captured at the cave; 2 more long-eared bats were briefly caught, but escaped. Other than this finding at this site, no other cave-related bat activity was detected below 550 m elevation.

Surface Habitat Use by Bats

Early Summer

Some bat use was documented (through ultrasonic detection and netting) at all elevational ranges. Bats were seen and detected flying along the littoral zone of Tahsis Inlet and the Leiner River estuary from February 1998 to 5 September 1998. In a quiet back eddy of the Leiner River, several *Myotis* sp. bats were detected based on call frequencies ranging from 50 to 80 kHz. *Eptesicus fuscus* was heard flying along the Leiner River (based on calls from 20 to 30 kHz). One lactating female *M. keenii* was captured at the Green Creek estuary on 8 July. The capture ratio for low elevation surface netting was 0.333 bats/net-night.

At mid- to high elevation, occasional feeding activity occurred in the forest/clearcut margins and marshes. The capture ratio in the mid- to high elevation surface was 0.1 bats/net-night.

Late Summer/Early Fall

As late as 4 September, calls consistent with *Myotis* bats were heard regularly over Tahsis Inlet and the adjacent Leiner estuary. Feeding activity occurred along the forest margins and marshes at high elevation but no bats were captured. Bats were also seen flying along trails in the forest and heard feeding over dolines, large sinkholes (>10 m deep) of solutional origin. At Weymer, they are typically devoid of trees within and provide open, uncluttered feeding zones within the forest.

Skull Identifications

All bat skulls were collected from cave passages above 730 m, except for 1 specimen (*Myotis* sp.) from Liquid Sky Cave at 380 m. This skull was coated with calcium carbonate, common to caves, which suggests an older specimen. It was not identifiable at the species level. *M. lucifugus, M. yumanensis, M. keenii, M. volans,* and possibly *M. evotis* were identified from skulls, including recently deceased (<2 years) individuals.

DISCUSSION

Temperature monitoring in caves showed that mean winter temperature declined with increasing elevation (Fig. 1A), whereas the coefficient of variation (CV) increased with elevation (Fig. 1B). However, the increased CV at higher elevation is due to variation from sites at cave entrances and outside caves (in Fig. 1B, all the points with low CV in higher elevation sites are from deep cave sites). Deep cave sites showed consistently low temperature variation (Fig. 2B, 3B).

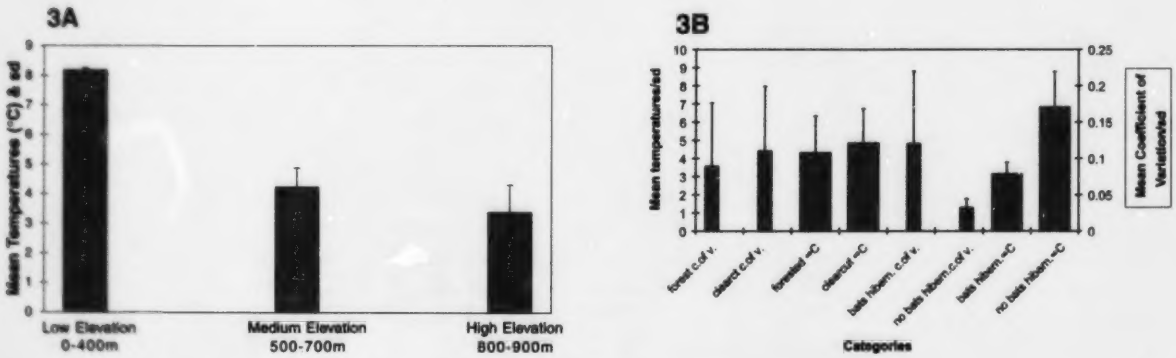

Figure 3. Bar graphs show differences in means and coefficients of variation (CV) of winter temperatures in deep cave sites, (A) at different elevations, and (B) with and without hibernating bats, under clearcuts and forest.

At low elevations, temperature variation was consistently low among cave depths, possibly due to the influence of the sea in some low caves, particularly one with a tidal sump. At mid- and high elevation the temperature variation increased nearer to entrances because of high temperature changes outside the cave and fluctuations in air flow direction between the surface and the stable deep cave environment. We found no apparent effect of clearcutting on winter temperature and variation in and near caves.

Our sampling showed that bats hibernate in Labyrinth, Wormhole, Deer Drop, Fracture, and Slot Canyon caves. These caves range in elevation from 592 to 843 m. They are all deep and long (>100 m). Their deep-cave mean winter temperatures range from 2.64 to 3.98°C , and the CV is very low (0.020–0.242). These conditions are best for long, undisturbed periods of torpor and low oxygen and fat consumption (McManus 1974, Tuttle and Stevenson 1977, Nagorsen and Brigham 1993).

No bats were found hibernating below 592 m elevation. Although deep, low elevation caves appear to have stable temperatures, the mean winter temperatures and metabolic costs may both be too high for hibernation (Hock 1951, Stones 1965, Davis 1970, McManus 1974, Tuttle and Stevenson 1977).

Bats use the entire study area at all times of year. The mid- to high elevation deep caves are used for hibernation by *M. keenii*, *M. volans*, *M. lucifugus*, and *M. yumanensis* from mid-October to mid-June, then for swarming from late July to late September. Low elevation riparian areas and littoral zones are used for feeding. At low elevation streams and Knoll Cave, we captured only female and young Keen's long-eared myotis. It appears that female *M. keenii* bats are feeding and raising young at low elevations and then moving to higher elevation caves for swarming and hibernation. This result suggests that the study area is very important to Keen's myotis for 2 critical aspects of its life history: rearing young and hibernation.

Climate has been shown to affect timing of reproduction in bats (Grindal et al. 1992), and probably affects timing of hibernation also. Thus, the late emergence of the 2 *Myotis* bats in Wormhole Cave may be due to a cool, wet spring. Likewise, the annual variation in swarming behaviour might be linked to weather patterns as discussed by Fenton (1969).

Bats use caves under both clearcuts and forests for hibernation and swarming. We noted that *M. volans* used a 15 X 15 X 25 m pit (Skypot) in a clearcut for feeding. This was the only recorded summer feeding use within a clearcut. However, we are not sure if the use of this cave for feeding is related to its unique thermal characteristics, which may be related to clearcutting, because we did not monitor temperatures here.

In summer, males predominate at higher elevation cave and surface sites. Females and young of the year appear with swarming males at the caves in mid-August. This difference in sexual distribution over an elevational and temporal range may be due to various physiological and abiotic factors such as variable energy demands between the sexes, prey availability, and climate. Females require warmer areas with greater insect abundance to meet the demands of raising young (Thomas 1988, Barclay 1991). Furthermore, the low number of bats documented feeding at high elevation bogs and marshes could be explained by a relatively wide distribution of more productive aquatic feeding areas at lower elevations. Green Creek and Leiner River at sea level, and Malaspina and Perry lakes at 100 m elevation provide good feeding areas. However, we do not know where bats roost during the early or late summer period.

Skeletal remains confirmed the presence of *M. keenii*, *M. volans*, *M. lucifugus*, and *M. yumanensis* in the caves. However, one of the *M. keenii* skulls collected has measurements that fall within the *M. evotis* range. To clarify taxonomic uncertainties, DNA is currently being sampled and analyzed. Until taxonomic uncertainties for field identification of *M. keenii* are resolved, we cannot identify the species with certainty (Firman and Barclay 1993).

At Weymer, diverse habitats at a wide range of elevations combined with deep caves create make for excellent conditions for hibernating, feeding, and raising young. We hope our hibernation model will be useful for locating other hibernacula on Vancouver Island.

CONCLUSIONS

1. Myotis bats hibernate in caves with stable temperatures between 2.4 and 4°C and 100% relative humidity. In this coastal montane environment, these are deep caves (>100 m long), and occur above 500 m elevation. The largest aggregations are found above 800 m.
2. Clearcutting appears not to affect winter temperatures deep in caves.
3. Myotis bats use the mid- to high elevation caves from mid-June to late July. Swarming occurs from late July to late September. Hibernation occurs from mid-October to mid-June.
4. Low elevation forest and riparian areas are used by Keen's long-eared myotis for feeding and raising young.

ACKNOWLEDGEMENTS

We thank the British Columbia Habitat Conservation Trust Fund and Bat Conservation International for funding this project in 1998. Forest Renewal British Columbia provided us with seed money for this study in 1996.

We acknowledge D. Nagorsen of the Royal British Columbia Museum for identifying bones collected from the caves and providing scientific direction and assistance. The

late S. van Zyll de Jong of the National Museum of Natural Sciences made the initial identification of the Keen's long-eared myotis at Weymer. J. Cook of the University of Alaska Museum is analyzing bat tissue DNA. We also thank the many members of the British Columbia Speleological Federation who have explored and mapped the Weymer Caves and field volunteers D. Nagorsen, K. Wright, M. Kissinger, M. Kaaremaa, R. Van den Dreissche, J. Joy, and S. Pendergast.

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Habitat Selection by Bats in Temperate Old-Growth Forests, Clayoquot Sound, British Columbia

Ruth van den Driessche[1]
British Columbia Ministry of Environment, Lands and Parks
2080-A Labieux Road, Nanaimo, BC, V9T 6J9, Canada

Trudy Chatwin
British Columbia Ministry of Environment, Lands and Parks
2080-A Labieux Road, Nanaimo, BC, V9T 6J9, Canada

Monica Mather[2]
7832 Plover Road, Lantzville, BC, V0R 2H0, Canada

ABSTRACT

Clayoquot Sound on the west coast of Vancouver Island, B.C. is an area of high-profile land-use conflict. In 1995, local scientific and management groups recognized bats as an important component of forest ecosystems. With almost no existing bat information for this area, we began studies of the bat community, with the goal of inventorying the bat community and identifying critical bat habitat at both landscape and stand levels. Identification of the important habitat will provide forest managers with guidelines that may minimize impacts of forest harvesting on bat populations. Over the 3 summers of field study we captured 6 species of bat: *Myotis lucifugus*, *M. yumanensis*, *M. californicus*, *M. evotis*, *M. keenii*, and *Lasionycteris noctivagans*. In addition, we confirmed the presence of *Eptesicus fuscus* through ultrasonic bat detection. From 1996 to 1998, Anabat II bat detectors were used to survey the relative flight and feeding activities of bat species in a variety of forest stands. Significantly higher bat activity was recorded in tall, open forests at lower elevations. In addition, more bat activity was recorded in stands dominated by amabilis fir compared to stands dominated by yellow-cedar. To evaluate the usefulness to bats of buffers around riparian areas, we placed bat detectors 10, 30, and 100 m away from creeks and lakes. There was significantly more foraging activity up to 30 m away from the water's edge, as compared to forest interior sites (>100 m). To assess roost sites as an element of critical habitat, we radio-tracked male and female bats to their roosts. We found 9 bat roosts. Although there are 6 main tree species in Clayoquot Sound, all 5 tree roosts were in large-diameter western redcedar trees, suggesting this is an important species for roosting bats. The remaining 4 roosts were found in cliffs.

Key words: bats, Clayoquot Sound, *Eptesicus fuscus*, habitat selection, *Lasionycteris noctivagans*, *Myotis*, temperate old-growth forest.

Among mammals of their size, bats are unique in having long lives, low reproductive rates, and relatively long periods of infant dependency (Findley 1993). In addition, bats may be dependent on forest attributes that are reduced in managed forests (e.g., large-diameter snags, riparian habitat, etc.; Kunz 1982, Crampton and Barclay 1998). These aspects of life history put forest-dwelling bats at risk of population decline in the presence of habitat alteration. In fact, half of the 16 species of bats found in British Columbia are threatened or endangered (Sarell and Luoma 1994). We know very little about bat habitat requirements for foraging and roosting in relation to old-growth forests (Nagorsen and Brigham 1993) and less still is understood regarding bats in temperate coastal forests. As the major predators of nocturnal flying insects (Rainey et al. 1992), bats play a key ecological role in temperate rain forests, therefore their distribution and habitat requirements are important to understanding and sustaining ecosystems in Clayoquot Sound.

In 1994, the provincial government appointed the Clayoquot Sound Scientific Panel in response to public concern regarding old-growth forest depletion. One recommendation from the panel was "to ensure that particular species known or suspected to be at risk are monitored and their habitats protected" (Clayoquot Sound Scientific Panel 1995). This inventory is a direct result of the panel's concern for the protection of key habitats, in particular for the Red-

[1] Present address: 598 Millstream Lake Road, Victoria, BC V9E 1E7, Canada
rvandena@sfu.ca
[2] Present address: 7017 Lavender Road, Lantzville, BC, V0R 2H0, Canada

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

listed Keen's long-eared myotis (*Myotis keenii*). No previous inventory of bats had ever been completed in any of the watersheds of Clayoquot Sound.

Beginning in 1996, efforts were made to sample bat populations of many undeveloped watersheds, with a few additional surveys in partially developed watersheds. Our objectives were to determine species presence and habitat preferences for foraging and roosting. This report summarizes the bats identified in Clayoquot Sound from 1996 through 1998, and presents and discusses habitat preferences of foraging and roosting bats, with greater emphasis placed on the results of 1998. This information is used to formulate our recommendations for managing bat habitat in Clayoquot Sound.

STUDY AREA

The study area is located in Clayoquot Sound on the west coast of Vancouver Island, B.C. (49°00'–49°30'N; 125°20'–126°30'W). The area is a maze of islands, inlets, and coastal temperate rain forest watersheds, and is classified as being in the Coastal Western Hemlock (CWH) biogeoclimatic zone and the Windward Island Mountains ecosection. This is the rainiest zone in British Columbia and is characterized by cool summers and mild winters (Meidinger and Pojar 1991). The area we sampled ranged in elevation from 0 to 730 m, but mountains in the area range up to 1,700 m. The vegetation of Clayoquot Sound consists of dense, coniferous forest with trees up to 60 m or more in height and 3 m or more in diameter. The understory of these forests is characterized by a dense tangle of shrubs, deadfall trees, and moss. The study area spans 3 biogeoclimatic subzone variants in the CWH zone (CWHvh1, vm1, and vm2), from sea-level on the outer coast to 730 m in inland watersheds.

In 1996 and 1997, we sampled in all 3 biogeoclimatic subzone variants. In 1998, we concentrated our sampling effort in the 2 lower-elevation subzone variants, where the most bat activity had been recorded in 1996 and 1997. Within each biogeoclimatic subzone we attempted to sample a range of aspects, habitat types, and forests of varying density and species composition.

MATERIALS AND METHODS

As there are inherent biases associated with different bat sampling methods (Thomas and West 1989), we used 3 methods to evaluate bat diversity and habitat preferences: mist-netting, monitoring of bat activity using bat detectors, and radiotelemetry.

Mist-netting

Our mist-net inventory followed the standardized Resources Inventory Committee protocol (RIC 1995). Each survey night from May through August we deployed 4–9 nets nightly for a minimum of 120 minutes post-sunset (the period of highest bat activity [van den Driessche et al. unpubl. observations]).

Bats were identified, weighed, and measured, and then released. In cases where identification was questionable, the 2 species names separated by a slash are used throughout this report (e.g., western/Keen's long-eared myotis).

Bat Activity

Measuring Bat Activity in Forest Interior Stands

The Vegetation Resource Inventory (VRI) system of maps in Clayoquot Sound is designed to provide a detailed inventory of forest vegetation and stand structure. Map polygon size averages about 8–10 ha. From the results of our work in Clayoquot Sound in 1996 and 1997, we identified 2 forest attributes (dominant canopy species and tree density) to help us "predict" which forest stands would have more bat activity. Based on these results, we selected VRI map polygons dominated by western redcedar (*Thuja plicata*), yellow-cedar (*Chamaecyparis nootkatensis*), western hemlock (*Tsuga heterophylla*), or amabilis fir (*Abies amabilis*), and polygons that varied in tree density in the CWHvh1 and CWHvm1 subzone variants. By placing ultrasonic bat detectors at stations in these polygons, we tested if, in fact, these mapped attributes could be strongly correlated to the amount of bat activity in the forest stand and therefore used as a guide in managing forests for bats.

Comparing Bat Activity in Riparian Stands to Activity in Forest Interior Stands

We wanted to evaluate bat activity in riparian areas with respect to riparian buffers in the Forest Practices Code of British Columbia. To compare bat activity in "riparian" versus "forest interior" sites, we chose sites that were 10 m and 30 m away from water at both lake and creek edges (riparian sites), or >100m away from water (forest interior sites).

Anabat Detector Protocol

For both the forest interior and riparian stations, the inventory technique was identical. We measured the relative abundance of bats using Anabat ultrasonic bat detectors at 42 forest interior stations and 43 riparian stations. We maintained a minimum of 100 m distance between detectors to ensure independence of samples (RIC 1995). Each site was sampled twice, once in May or June, and again in July or August.

Cassette tapes of echolocation calls were analyzed by ear the morning after the survey. As almost all calls recorded were *Myotis* species, we combined all bat species and recorded the total number of echolocation passes per night.

Habitat Plots to Compare Forest Stands

At each Anabat detector station we sampled 20 X 20 m plots

to compare actual measures of vegetation and stand characteristics to the VRI map, and also to measure the habitat elements we believed might influence bat activity.

We classified every tree >10 cm DBH (diameter at breast height) in the plot to a standard stratum, and identified species and decay class (British Columbia wildlife tree classification scheme; Vegetation Inventory Working Group 1994). We noted height of canopy, elevation, and slope.

Radiotelemetry to Locate Roosts

In 1997 and 1998, we radio-tracked 9 bats (mass >5.7 g) fitted with Holohil LB-2 transmitters and flexible antennae (Holohil Systems Ltd., Carp, ON). To increase efficiency, since much of the terrain is extremely steep and thick with undergrowth, we used a fixed-wing aircraft to establish the general area and then retraced the distance on foot to the bat roost.

Once the roost was found, we took measurements of the roost structure and surrounding habitat. We characterized the forest stand surrounding the tree roost in a 20 X 20 m habitat plot in the same manner as at the Anabat detector sites.

Statistical Analysis

We averaged the number of bat calls per hour to use as a uniform measure of relative abundance (or bat activity) and then did a log transformation to equalize the variance. The number of passes detected per unit time can be considered as an index of the relative use or abundance at different sites and thus as an index of habitat preference (Thomas 1988).

Detector stations were classified as either riparian (at 10 and 30 m from water) or forest interior (>100 m from water) stations. In the analysis of habitat variables (forest density, canopy height, slope, and elevation), we used only forest interior detector sites to remove the compounding effect of riparian habitat from the model.

As we visited each detector station twice, we used a nested (repeated measures) 1-way analysis of variance (ANOVA) to determine if numbers of bat passes/hour varied for class variables such as biogeoclimatic subzone variant, dominant canopy species (measured in the field), and VRI dominant tree species (SAS 1988). We also used the nested (repeated measures) ANOVA to determine if numbers of bat passes differed significantly between stations that were 10 and 30 m from water and stations in the forest interior. To test if habitat variables (e.g., forest density, etc.) differed between stands dominated by different tree species, and between riparian and forest interior sites, we used an ANOVA.

We tested all continuous variables for normality and log-transformed the data to equalize the variance where needed (Zar 1984). If variables could not be normalized, we did not include them in the regression models. We used simple linear regressions to evaluate effects of continuous variables on relative bat abundance. We included a class variable for site in all regression models to account for multiple site visits. As a result of this, there was a large amount of variability inherent in the model and this was reflected in the low R^2 values. We chose to use simple linear regression on the variables rather than a correlation matrix in order to include the variability within each detector site (using both visits to each station) and to reduce the chance of making a Type I error. However, where the variances could not be normalized we used correlations to evaluate relationships (Spearman's rank correlation).

RESULTS

Bats Captured Using Mist-nets

From 3 July to 27 September 1996, 19 May to 22 August 1997, and 19 May to 30 August 1998, we set 753 mist-nets and caught 118 bats. We caught 1 lactating female western long-eared myotis (*Myotis evotis*; mass outside range of Keen's), 1 male Keen's long-eared myotis (*M. keenii*; from external measurements, Dave Nagorsen, Royal British Columbia Museum, pers. comm.), and 7 other western/Keen's long-eared myotis. Fifteen California myotis (*M. californicus*) were captured. We positively identified 3 Yuma myotis (*M. yumanensis*) and 3 little brown myotis (*M. lucifugus*; based on forearm measurements), and caught 53 little brown/Yuma myotis. In May 1998, we caught 2 silver-haired bats (*Lasionycteris noctivagans*) in an estuarine meadow at Cow Bay, Flores Island. We confirmed presence of reproductive females in the populations of western long-eared myotis, California myotis, Yuma myotis, and little brown myotis. Using ultrasonic detectors we identified an additional species (big brown bat [*Eptesicus fuscus*]), which was not captured in the mist-nets.

Comparing Bat Activity in Different Forest Stands

We compared bat activity in the 3 biogeoclimatic subzone variants of Clayoquot Sound (CWHvh1, CWHvm1, CWHvm2). Among the 3 subzone variants, significant differences in bat activity were detected (nested ANOVA; df = 2, $F = 6.8$, $p < 0.05$) with significantly more bat activity in CWHvh1 than in CWHvm2 (Scheffe's pairwise comparisons, $p < 0.05$).

We selected VRI polygons dominated by 4 tree species (western redcedar, yellow-cedar, amabilis fir, and western hemlock) and looked for differences in bat activity between them. We found no difference between the 4 mapped stand types (nested ANOVA, df = 3, $F = 1.94$, $p > 0.05$). However, we did find significant differences in activity between stands where the dominant species was determined from our 20 X 20 m habitat plots (nested ANOVA, df = 3, $F = 3.89$, $p < 0.05$). Yellow-cedar stands had significantly less bat activity than amabilis fir stands (Scheffe's test, $p < 0.05$). When we compared the VRI-mapped dominant tree species to the dominant canopy species from habitat plots, they corresponded 50% of the time.

As bat activity was thought to be influenced by stand density and canopy height, we compared these variables

between stands that differed in dominant tree species and found stands dominated by yellow-cedar were shorter and more dense than other stands (Table 1). When we compared bat activity with the species' proportion in a plot, we found less bat activity in stands with higher proportions of yellow-cedar (correlation; $n = 43$, $R = -0.34$, $p < 0.05$).

Consistent with results of 1996 and 1997 (linear regression, df = 55, $F = 5.99$, $p < 0.05$), we found bat activity significantly lower in denser forest stands. Variables that best described this relationship were the total number of trees, number of upper subcanopy trees, and canopy height, all of which are measurements that were taken in the field (Table 2). VRI estimates of forest density (number of tree stems/ha and canopy closure) did not exhibit the same negative regression to bat activity. However, when we examined the relationship between measured and VRI-mapped density variables, we found that measured density did not correlate with the VRI estimate of this same attribute ($n = 79$, $r = 0.13$, $p > 0.05$). Results also suggested that height of canopy and elevation were significant factors in determining bat activity.

Comparing Riparian Sites to Forest Interior Sites

In 1997, we compared *Myotis* activity over water to *Myotis* activity in the forest, and found significantly more activity over water (paired t-test, df = 33, $t = 6.6$, $p < 0.0001$). In 1998, we compared bat activity in riparian stands (10 and 30 m from S2 creeks and lakes) and forest interior stands (>100 m from water). Riparian sites had significantly more activity than forest interior sites (nested ANOVA, df = 2, $F = 4.35$, $p < 0.05$). Although there was not significantly more activity at the 10-m sites than the 30-m sites, the mean number of bat passes per hour suggested that bat activity did drop as the detectors moved away from water ($\bar{\chi}_{10m} = 1.51$, u.c. = 0.39, $\bar{\chi}_{30m} = 1.38$, u.c. = 0.33, $\bar{\chi}_{Forest\ Interior} = 1.00$, u.c. = 0.23).

We investigated differences in stand structure between 10-m riparian, 30-m riparian, and forest interior sites to see if this could provide some insight as to why bats were selecting riparian areas for foraging and flying. We tested variables that had shown a significant relationship to bat activity at forest interior stations (tree density, upper subcanopy, and canopy height) and found no difference in any of these habitat variables (ANOVA, df = 2, $F = 2.25$, 0.88, 0.05, respectively, $p > 0.05$).

Table 1. Comparison of stand density and canopy heights of stands dominated by 4 species of tree. (ANOVA, df = 3, all variables are significant. Scheffe's test is a multiple comparison test that determines which variables are significantly different from others.)

Variable name	*F*-value	Scheffe's test results[a]
No. of trees in plot	6.84	YC > WRC, WH
No. of upper subcanopy trees	4.00	YC > WRC, WH
Canopy height	10.89	YC < WRC, WH, AF

[a] YC = yellow-cedar; WRC = western redcedar; WH = western hemlock; AF = amabilis fir.

Radiotelemetry to Locate Roost Sites

We located 9 bat roosts. All bats found roosting in trees were found in western redcedars (Table 3). We found tree-roosting bats in large-diameter (115–315 cm DBH) western redcedar trees and most frequently in trees of decay class 2 (4 out of 5 bats). The remaining bats were found in steep, inaccessible rock faces with the exception of 1 *M. lucifugus* male roosting in a shallow pocket of a seacliff just 5 m above the spray zone.

DISCUSSION

Bat Species Diversity in Clayoquot Sound

Prior to 1996, only the Yuma myotis and little brown myotis had been identified in the CWH biogeoclimatic zone of

Table 2. Results of linear regression analysis comparing bat activity to stand characteristics of forest interior stations. Significant regressions and correlations have asterisks. Correlation coefficients for VRI (Vegetation Resource Inventory) canopy height and elevation are Spearman's rank correlations.

Variable class	Independent variable[a]	Degrees of freedom	*F*-value	Relationship
No. of stems	All trees	42, 1	8.54*	$R^2 = 12.4\%$ (-)*
	Canopy trees (L.T.)	42, 1	1.51	
	Upper subcanopy trees (L.T.)	42, 1	7.07*	$R^2 = 10.6\%$ (-)*
	Lower subcanopy trees	42, 1	3.44	
	VRI tree density (L.T.)	38, 1	0.76	
Canopy height	Estimated canopy height	36, 1	5.99*	$R^2 = 9.9\%$ (+)*
	VRI canopy height	$n = 39$		$r_s = 0.24$ (-)*
Physical Characteristics	Slope	42, 1	1.91	
	Elevation	$n = 38$		$r_s = 0.59$ (-)*

[a] L.T. indicates log transformed data.

Table 3. Summary of summer bat roosts found using radiotelemetry in Clayoquot Sound. The first 2 individuals in the table switched roosts and both roosts are listed. All radio-tagged female bats were lactating.

Species[a]	Sex	Elevation (m)	Slope (%)	Aspect (°)	Roost structure[b]	Tree height (m)	Tree diameter (cm)	Wildlife tree class
MYLU/YU	M	30	17	80	WRC	32	220	2
		35	47	230	WRC	35	120	2
MYLU/YU	M	40	20	240	WRC	17	115	2
		0	90	225	sea cliff			
LANO	M	10	0	flat	WRC	26	225	2
MYLU/YU	F	10	0	flat	WRC	37	315	6
MYLU/YU	F	900	sheer	160	cliff			
MYLU/YU	F	600	sheer	225	cliff			
MYLU/YU	M	475	sheer	140	mixed tree/cliff			

[a] MYLU/YU = *Myotis lucifugus/yumanensis*; LANO = *Lasionycteris noctivagans*
[b] WRC = western redcedar; sea cliff = cliff adjacent to ocean, just above spray zone; cliff = cliff located inland or not adjacent to ocean.

Vancouver Island (Firman et al. 1993). Our study confirmed the presence of 5 other species: the endangered Keen's long-eared myotis, western long-eared myotis, California myotis, silver-haired bat, and big brown bat. The presence of pregnant and lactating western long-eared myotis, California myotis, Yuma myotis, and little brown myotis indicates that breeding populations of these species are present in the area.

The presence of the endangered Keen's long-eared myotis in Clayoquot Sound is of special significance as it had been found previously at only 4 other Vancouver Island locations (Nagorsen and Brigham 1993). It is difficult to determine the distribution or population of the species in Clayoquot Sound since mist-netting is biased to low-flying species (Thomas and West 1989). We caught fewer than 8 Keen's long-eared myotis out of 118 bats, which suggests this species is rare; however, further studies using methods more suited to sampling Keen's long-eared myotis are needed to confirm the species is indeed rare in our study area.

Comparing Bat Activity in Different Forest Stands

Comparison of Biogeoclimatic Variables

Relative numbers of bat passes differed between the 3 CWH subzones. The lowest elevation detector sites on the outer coast of Clayoquot Sound (CWHvh1) had the highest numbers of bat passes, compared to the inland, high-elevation sites in the CWHvm2. Inland and higher elevation sites are exposed to higher amounts of precipitation, longer winters, and greater extremes of daily and annual temperatures (Meidinger and Pojar 1991). We suggest that bats are less abundant, as greater amounts of precipitation and cooler temperatures will significantly affect prey abundance and foraging ability (Grindal et al. 1992).

Comparing Density of Forests

Over 3 years of research we observed a negative relationship between bat passes and stand density, supporting the idea that forest density affects the foraging and flying behaviours of bats and thus their distribution (Fenton 1990). Low-density forests can be viewed as containing fewer obstacles or "clutter." Cluttered habitats are harder for bats to efficiently echolocate within due to multidirectional reflections of their calls, and harder to manoeuvre within due to morphological constraints (Neulweiler 1984, Mackey and Barclay 1989). Forest openings determine forest structure (Arsenault and Bradfield 1995) and may be critical factors in defining bat foraging habitat at the microhabitat scale. The density of the upper subcanopy trees was negatively correlated to bat activity, suggesting that gaps in these forests exist both vertically and horizontally. As bats can fly, this allows movement into different vertical microhabitats by choosing to fly lower or higher (Bradshaw 1996). Forest openings in structurally diverse, old-growth forests represent relatively uncluttered commuting "pathways" and foraging areas, and may be extremely important to bats (Bradshaw 1996, Krusic and Neefus 1996).

Comparing Stands Dominated by Different Canopy Species

We found a significant difference in bat activity between the 4 dominant canopy species measured in a habitat plot. Stands of yellow-cedar had lower bat activity than stands of amabilis fir. We found a positive relationship between yellow-cedar and tree density, and significantly less bat activity in forests with relatively high representation of yellow-cedar. As yellow-cedar-dominated forests are typically denser and shorter, this may explain why we found less bat activity in these forests.

The discrepancy in results between measured (significant) and mapped (nonsignificant) habitat variables was unexpected. However, dominant canopy species of habitat plots were assigned from data collected in a 20 X 20 m plot and the VRI species was based on an 8- to 10-ha polygon area. Therefore, inconsistent results could be due to the different scale of assigning dominance. As bats did appear to respond to dominant canopy species at the 20 X 20 m plot scale, we

suggest that site investigations at this scale are preferable for identifying important habitat for bats.

Riparian Areas

Our data suggest riparian areas are extremely important to bats. As the distance increased from water, bat activity decreased. As we looked at habitat characteristics (which limit manoeuvrability and echolocation efficiency) between riparian sites and forest interior sites and found no significant differences, it is probable that bats are selecting riparian areas in response to other factors, such as higher insect abundance. Therefore, bats are selecting to forage in riparian areas for different reasons than they are selecting forest interior habitat, where stand structure did have an influence on their flight paths. The activity patterns of insectivorous bats and insect abundance have been shown in some studies to have a clear relationship (e.g., Rautenbach et al. 1996). Therefore, the 50-m riparian buffer proposed by the Clayoquot Sound Scientific Panel may better protect this habitat than the 30-m Riparian Reserve Zone legislated in the Forest Practices Code.

ROOST SITES IN CLAYOQUOT SOUND

We confirmed the use of cliff bands and large-diameter cedar trees as roosts for both male and female bats. In other studies, bats preferred large-diameter trees for roosting (e.g., Brigham 1991, Crampton and Barclay 1998), and timber harvesting has been associated with a decline in abundance of bats when large-diameter trees are cut (Lunney et al. 1985).

We found 4 of 5 roosting bats in western redcedar trees in a particular stage of decay. Typical tree roosts had dead tops and were in declining health (candelabra cedars in wildlife tree class 2), with extensive cracks leading to interiors decayed by heartrot. As trees beyond wildlife tree class 6 no longer provide suitable roosting habitat for bats (Brigham et al. 1997), preserving large-diameter cedar trees at early stages of decay (class 2) will protect roost sites for bats over a long period, since cedars decay slowly (Parish 1994).

Roost switching has been observed for several species of temperate bats (Brigham 1991, Vonhof and Barclay 1996, Brigham et al. 1997). We documented male bats switching roosts, both between trees and from tree to cliff. Our observations and the literature suggest that preserving a series of potential roost sites may be important for maintaining populations of bats in Clayoquot Sound.

SUMMARY OF MAIN FINDINGS

To maintain populations of bats in Clayoquot Sound, forest managers will need to consider roosting and foraging requirements at the stand and landscape levels. Seven species of bats were confirmed for the area, including the endangered Keen's long-eared myotis, western long-eared myotis, California myotis, Yuma myotis, little brown myotis, silver-haired bat, and big brown bat. The presence of pregnant and lactating western long-eared myotis, California myotis, Yuma myotis, and little brown myotis indicates that breeding populations of these species are present in the area.

AT THE STAND LEVEL

1. We observed a negative relationship between bat passes and stand density, supporting the idea that denser forest is less attractive for foraging and flying bats.
2. We confirmed the use of cliff bands and large cedar snags as roosts for both male and female bats. Tree roosts were predominately located in large-diameter western redcedar trees in early (class 2) decay stages. Thus, preserving these trees will ensure their long-time use as roost sites for bats. We documented male bats switching roosts, both between trees and between tree and cliff. This implies that a series of potential roost sites should be preserved to maintain populations of bats in Clayoquot Sound.
3. We found significantly more bat activity in stands dominated by amabilis fir and less activity in stands of yellow-cedar.

AT THE LANDSCAPE LEVEL

4. Our data suggest riparian areas are extremely important to bats. We found that bat activity at 10 and 30 m from water was significantly more than in forest interior stands (>100 m from water). Current riparian zones legislated in the Forest Practices Code protect a 30-m Riparian Reserve Zone around S2 creeks and 10 m around L1 and L2 lakes. The Clayoquot Sound Scientific Panel recommendations have proposed a 50-m buffer around these same areas. Our data suggest a buffer >30 m on creeks and lakes may better protect this habitat for bats.
5. Bat activity differed between the 3 subzones of the Coastal Western Hemlock biogeoclimatic zone. The lowest-elevation detector sites on the outer coast of Clayoquot Sound (CWHvh1) had the highest numbers of bat passes, compared to the inland, low-elevation sites in the CWHvm1 and the inland, high-elevation sites in the CWHvm2.

ACKNOWLEDGEMENTS

Forest Renewal British Columbia granted funding for the project. We are indebted to H. Bury, E. Grundmann, T. Luszcz, C. Sutherland, L. Crampton, M. Meredith, K. Allard, and dedicated volunteers for their assistance and good humour in the field. L. Jones provided advice, guidance, and administrative support. We thank J. Joy and A. Eriksson for their improvements to an earlier version of this report.

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Bats of Gandl K'in

Douglas W. Burles
Gwaii Haanas National Park Reserve/Haida Heritage Site
P.O. Box 37, Queen Charlotte City, BC, V0T 1S0, Canada
Doug_Burles@pch.gc.ca

ABSTRACT

Gandl K'in, Haida Gwaii (Queen Charlotte Islands) supports an unusual maternal colony of Keen's long-eared myotis and little brown myotis. Hot water from geothermal springs flow through crevices and under boulders creating warm, moist conditions for roosting bats. Some boulder roosts are unique in that they are in the upper intertidal zone and are periodically flooded by extreme tides. The pattern of seasonal use strongly suggests that the thermally heated roosts are important during pregnancy and lactation, but the bats roost elsewhere once the young are volant. Emergence counts confirmed that at least 110 adult bats used the Gandl K'in roosts during 1998, which is similar to the 140 adults and young estimated in 1991. The 2 species were present in approximately equal numbers in both 1991 and 1998, which suggests they are coexisting with minimal competition. Roost sites do not appear to be limiting, and the preliminary results of feces analysis suggest that different foraging behaviour may be minimizing competition for food.

Key words: hot springs, Keen's myotis, little brown bats, *Myotis keenii*, *Myotis lucifugus*, roosts.

The Haida call them "Guudagiigumhlgahl"—the animal that hangs upside down. Bats may be one of the most abundant mammals in North America. Their ability to fly has enabled them to greatly diversify and to colonize virtually all tropical and temperate regions of the world. The evolution of an advanced echolocation system has allowed them to navigate and forage in total darkness, and thus occupy niches with few competitors or predators.

While the abilities to fly and to echolocate have allowed bats to greatly diversify, they have also imposed significant energy constraints. These abilities require large amounts of energy to maintain, and these costs, especially when coupled with very small body size, make it difficult for bats to maintain a positive energy balance. Bats occupying temperate regions have thus had to evolve both physiological and behavioural strategies to survive. Most have become heterothermic, and use torpor to conserve energy. But torpor delays fetal development, so it is not a good strategy for reproductive females. Instead they seek out day roosts that will provide an external source of heat that will assist in maintaining body temperature. The hot springs of Gandl K'in provide just such roosting conditions.

Sixteen species of bats are known to occur within British Columbia, 9 of which occur along the north coast mainland. Only 4 species have been found on Haida Gwaii (Queen Charlotte Islands), perhaps a reflection of the islands' remote nature. Silver-haired bat (*Lasionycteris noctivagans*), little brown myotis (*Myotis lucifugus*), and California bats (*M. californicus*) are relatively common, but the fourth, Keen's long-eared myotis (*M. keenii*) is rare.

Keen's myotis has one of the most restricted distributions of any bat in North America, a distribution that may be restricted to mature, temperate coastal rain forests of the Pacific Northwest. First collected near Massett on Haida Gwaii in 1895, it has subsequently been found only rarely along the coast of British Columbia, Washington State, and southeast Alaska. Between 1895 and 1960 only 10 specimens were collected from various locations along the coast of British Columbia (D. Nagorsen, Royal B.C. Museum, Feb 1998, pers. comm.). During the early 1960s a number of Keen's myotis, including juveniles, were collected at Gandl K'in (Hotsprings Island) by Foster (1965). Extensive surveys conducted throughout British Columbia during 1991 documented Keen's myotis at only 1 location, Gandl K'in (Firman et al. 1993). Because of the rarity of this species, its restricted distribution, and its uncertain population status, this species is on the British Columbia Red List of species of concern, and is listed as vulnerable by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC).

STUDY AREA

Gandl K'in (Haida for "healing waters") is a 15-ha island located about 15 km east of Moresby Island on Haida Gwaii, which is approximately 100 km off the northwest coast of British Columbia. It lies within the Coastal Western Hemlock wet hypermaritime biogeoclimatic subzone.

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However, because of its location in the rain shadow of the San Christoval Mountains and its southerly aspect, it is relatively warmer and drier than adjacent islands. Two thermal meadows occur where the heat of the springs appears to be preventing trees and shrubs from encroaching. Numerous small warm water seeps occur within and around the edge of these meadows. Water temperatures of up to 77°C have been recorded at some of these seeps (Sadlier-Brown 1994). Bats take advantage of the moist, heated crevices and cavities, including some that are in the upper intertidal zone, to roost and raise young.

Archaeological evidence has shown that the site has been used traditionally for at least the last 2,000 years (D. Fedje, Parks Canada, pers comm., 1998). Historically, the site has been a popular destination; as a result, many facilities have been developed ad hoc. It continues to be one of the most popular locations within Gwaii Haanas, with up to 3,000 visitors stopping each summer to enjoy the hot pools.

RESULTS

Roost Descriptions

Bats were found to roost in 3 areas spread over about 100 m around the hot springs. Roosts were located primarily by observing emergence and by tracking radio-tagged bats. Most bats roosted under large boulders in a boulder field located about 40 m down the beach from the bath house. Numerous passages and galleries occur under these boulders, and the hot water that seeps through them creates warm, moist areas to roost. Water temperatures up to 50°C and air temperatures up to 33°C were recorded in some of these passages and galleries. Most roosts identified here were above the extreme high tides but one roost, used by a radio-tagged little brown bat, was later flooded by extreme tides.

Bats also regularly roosted in a crevice in the cliff further down the beach. This crevice is up to 75 cm deep and is thermally heated by seepage from the bathing pool directly above it. Temperature gradients generally occur within the crevice, with temperatures reaching 32–34°C in the innermost reaches of the crevice, and grading down to the low 20°C near the outer edge. Ambient temperature and cloud cover also influenced temperatures within the crevice. On particularly warm, sunny days, temperatures near the outer edge were similar to or higher than those of the central regions (Fig. 1). Bats using this roost could thus move around to find the most desirable temperature. On 5 occasions the temperature where little brown bats were seen in this crevice was determined using a thermometer with a remote sensor. These temperatures ranged between 27 and 33°C. Bats were occasionally seen in other adjacent cracks in this cliff (K. Hans, Haida Gwaii Watchmen Program, pers comm., July 1998), although most other cracks are not thermally heated.

Bats were also found to roost in thermally heated rocks directly above the main bathing pool. This roost was accessed through a 1 m hole in the salal (*Gaultheria shallon*) cover and consisted of a 3–5 cm wide crevice. At least 18 bats were seen to emerge from this roost on 23 May 1998.

Colony Size

The first estimate of the size of this colony was determined in 1991 by Firman et al. (1993). Using mist netting and emergence counts during late July–early August, they estimated that at least 70 Keen's myotis and at least as many little brown bats were using these roosts. These numbers include volant juveniles and nonreproductive males and females, as well as breeding females.

Roost emergence counts were conducted regularly during 1998 to determine use patterns and numbers of bats using the roosts. Because the bats roost in several locations up to 100 m apart around the hot springs, it was necessary to first determine roost sites and emergence patterns using a number of observers. Once the preferred flight corridor was determined, a location was chosen where emerging bats could be seen against the sky and a single observer could most effectively conduct emergence counts.

Formal counts were scheduled to coincide with periods of moderate and extreme tides to determine tidal influence on the use of roosts. They were usually carried out on 2 consecutive nights to determine nightly variation in use. A bat detector was used during all counts to supplement observations.

A total of 23 formal counts were conducted over the summer (Fig. 2A). Early in the season relatively few bats were using the thermally heated roosts. A dead bat was found in March (I. Wilson, Haida Gwaii Watchmen Program, pers comm., June 1998), suggesting some early season use, but no bats were seen during preliminary study surveys on 12 and 13 May (this study). Systematic counts began on 23 May with 21 bats being counted. Numbers increased dramatically during late May–early June with the highest count of the summer (110) being made on 29 May ($\bar{\chi}$ = 42, range = 3–110, N = 7). During late June–early August (the period of late pregnancy and parturition), counts were similar, averaging 46 bats per count (range 15–87, N = 10). Numbers decreased dramatically during mid- to late August, with a mean of only 11 being counted (range 1–35, N = 4). A count on 24 September revealed that 1 bat was still using the Gandl K'in roost.

The high number of 110 bats is probably the best estimate of the number of females of both species using this maternity colony, as all bats captured at Gandl K'in during May–June were adult females. This figure is not directly comparable with the estimate by Firman et al. (1993) because the counts were taken at different times of the year; however, it does confirm the relatively small but stable size of the colony.

Why Such Variation in Counts?

Weather significantly influences bat activity, and was a factor in the variable use of the Gandl K'in roosts. On 7 June, for example, weather conditions were optimal and 61 bats emerged. However, on the next evening, the sky was overcast, and rain threatened; only 28 bats emerged. Around 2345 on 8 June, a steady rain began to fall and continued until 1200 the next day. This rain seems to have prevented bats from returning to the roosts as only 3 bats emerged the following night. Similarly, on 6 August a misty rain persisted throughout the evening, and only 1 bat was seen to emerge.

Early investigations of the Gandl K'in colony indicated that the roosts were intertidal, and were thus periodically abandoned during extreme tides (Firman et al. 1993). One intertidal roost was identified during this study, but most were supratidal. Emergence counts were compared with the daytime high tide level (Fig. 2B), and with the state of the tide at approximately the time that bats would be returning to the roosts (Fig. 2C). No correlations could be identified; in fact, the highest count of the summer occurred when the lower roosts were being flooded daily.

Seasonal use of the roosts accounts for much of the variation in count numbers. Few bats used the roosts early in the season, and then only irregularly. Many bats arrived in late May, but not all remained there. Numbers using the roosts remained high during June and early July (period of pregnancy and early post-parturition), and increased during mid-July, probably a reflection of onset of volancy by young and the arrival of nonreproductive females and males into the area. The first male, juvenile, and nonreproductive female were captured on 19, 20, and 31 July, respectively. Numbers using the roosts began to decrease by late July, as post-lactating females and volant young abandoned the roosts for more favourable locations. By late August and September, only a few bats were still using the roosts.

Incomplete coverage during counts may account for some of the variability. Because the roosts were spread over about 100 m, all exits could not always be monitored. A protocol

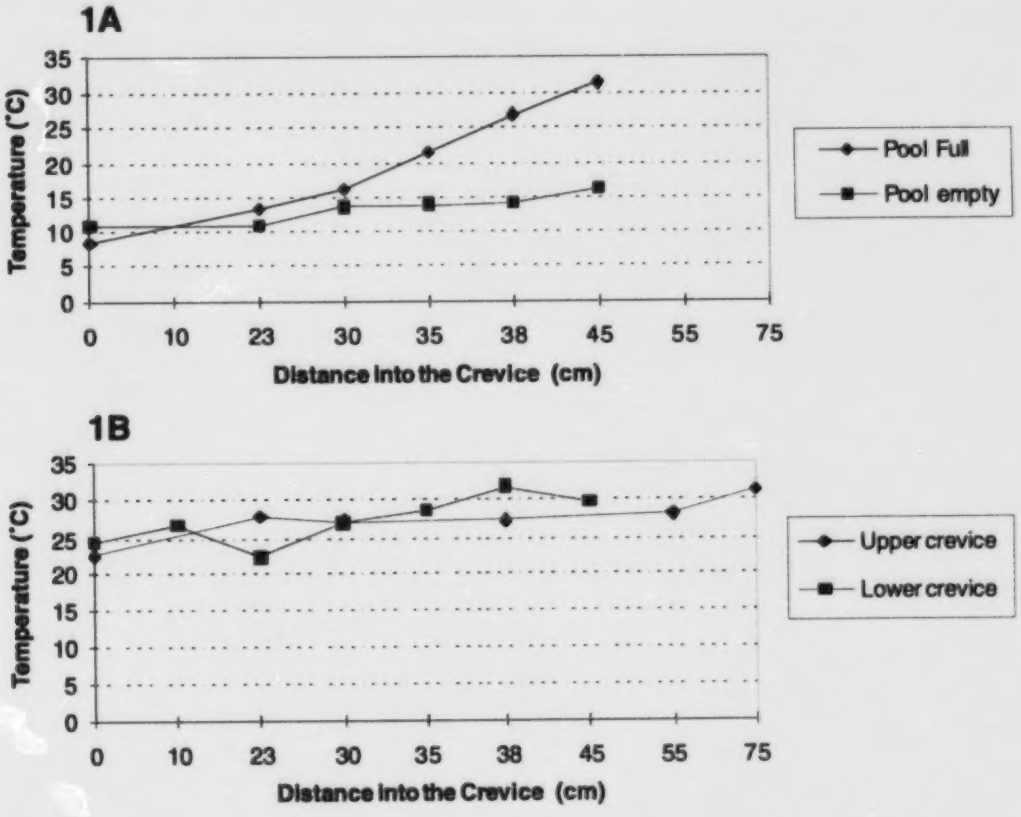

Figure 1. Temperature gradients observed in a crevice used by bats for roosting, Gandl K'in, Haida Gwaii. 1A demonstrates the importance of warm water seepage on crevice temperature, 22 Feb 1998; while 1B demonstrates the importance of radiant heat on crevice temperatures 1 July 1998, when ambient temperature was 22–24°C.

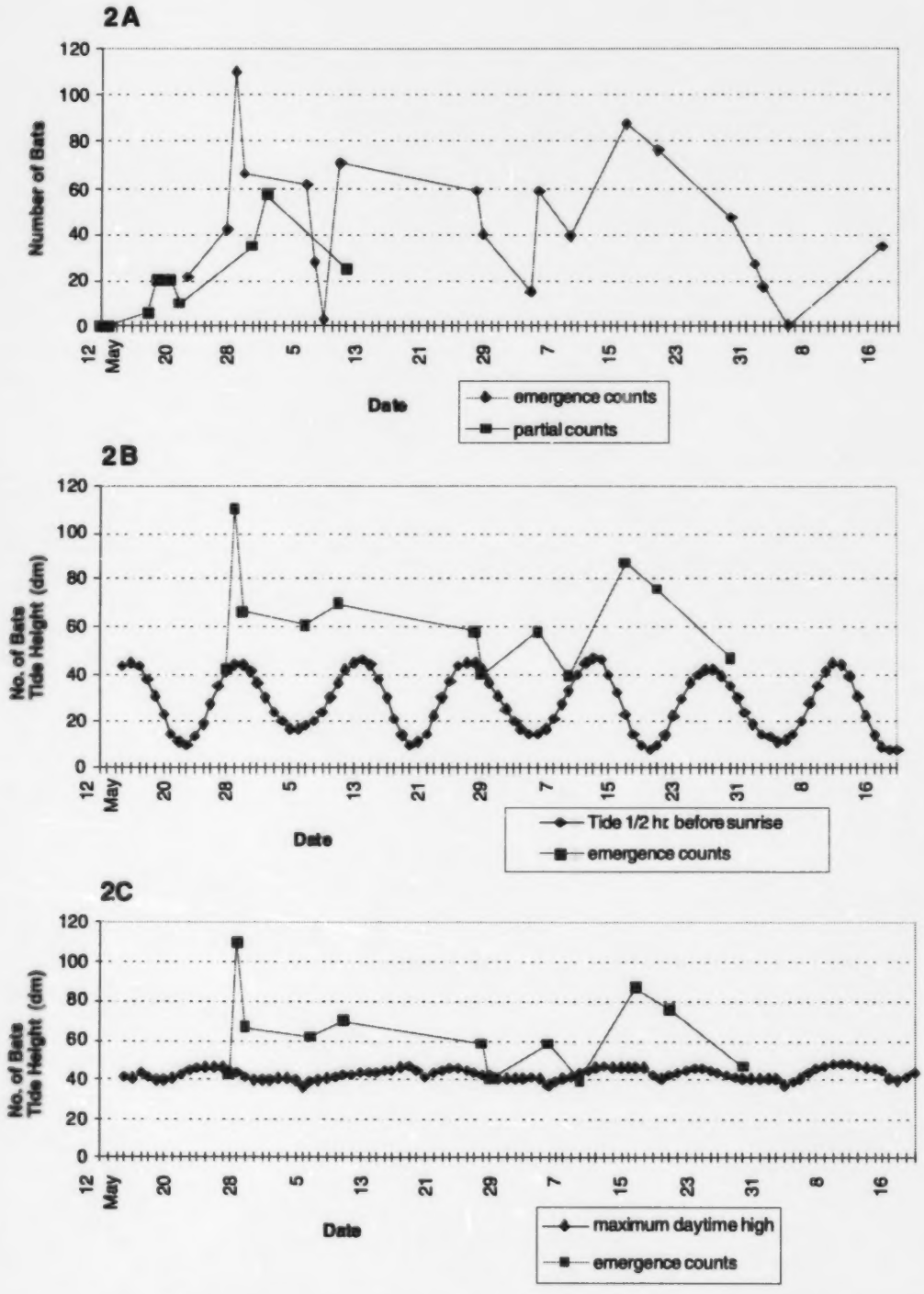

Figure 2. Summary of emergence counts taken at Gandl K'in, Haida Gwaii during 1998 and their relation to tide height. 1A: Emergent vs. partial counts. 1B: Emergent counts vs. tide height 1/2 h before sunrise. 1C: Emergent counts vs. maximum daytime tide height.

Table 1. Summary of classified counts of *M. keenii* and *M. lucifugus* at Gandl K'in 1998.

Date	Total no. counted	No. of *M.keenii*	No. of *M.lucifugus*	Percent *M.keenii*
18 May	6	3	3	50.0
28 May	42	3	3	50.0
29 May	110	18	17	51.4
2 June	57	23	23	50.0
3 June	42	13	6	68.4
7 June	61	11	16	40.7
8 June	28	13	9	59.1
28 June	58	11	9	55.0
5 July	15	3	4	42.9
6 July	58	21	5	80.8
17 July	87	17	26	39.5
21 July	76	2	3	40.0
				$\bar{\chi}$ = 52.3

was established to monitor the preferred emergence route. Some counts were conducted using multiple observers and/or automated bat detectors to monitor all exits from the roosts. These counts confirmed that not all bats were using the flight corridor identified, but the small numbers of detections (up to 7) could not explain all of the variation observed.

What Species Use Gandl K'in?

Although 4 species have been recorded on Haida Gwaii, only Keen's myotis and little brown bats have been captured on Gandl K'in (Firman et al. 1993, this study). In 1991, Firman et al. captured 77 bats, 29 (37.7%) of which were Keen's myotis. During the first year of this study, 67 bats were captured on Gandl K'in, 24 (35.5%) of which were Keen's myotis. These ratios must be used cautiously, however, as one species may be more susceptible to capture than the other.

A second method for establishing a ratio of the 2 species present was developed based on differences in the strength of their echolocation calls. The more intense call of little brown bats was detectable at a range of about 10 m while the lower intensity call of Keen's myotis was only detectable at about 3 m. For bats that could be seen as they approached, the distance of first detection could be determined. Keen's myotis also tend to fly erratically and very low and close to vegetation, while little brown bats usually fly in a straight line 3–5 m above the vegetation.

A total of 262 sightings were classified to species, 52.3% of which were Keen's myotis (Table 1), a ratio similar to that proposed by Firman et al. (1993). Using this figure, it is estimated that at least 58 female Keen's myotis used the Gandl K'in roosts in 1998. The consistency in ratios between 1991 and 1998 suggests that these 2 species are coexisting with little or no competition for roost sites.

How Do These Two Species Coexist?

Whenever 2 similar species are found together, their ability to coexist depends on their level of competition for resources. If there is a high degree of competition, the superior competitor should eventually outcompete, and thus displace, the inferior one. The 2 species in this study are of the same genus, have similar roosting requirements, and are both insectivores. The data show that they have coexisted in almost equal numbers at Gandl K'in for the last 8 years, and from all appearances are not competing for roosts.

Two species with similar diets might be expected to forage in a similar manner and thus compete for food. As part of this study, feces samples were collected from all captured bats for diet analysis. Table 2 provides a preliminary summary of arthropod remains found in scats of Keen's myotis (N = 13) and little brown bats (N = 11). Both species appear to be feeding largely on Lepidoptera, although this may be misleading, as in all cases only a small number of scales (usually less than 15) were found. The remainder of their diets differs. The relatively high proportion of arachnids in the diet of Keen's myotis suggests that these individuals may be deriving a considerable part of their diet by gleaning spiders from vegetation. The high proportion of dipterans and neuropterans in the diet of little brown bats is consistent with the identification of this species as an aerial insectivore. These results, while preliminary, suggest that diets only partially overlap, and different foraging behaviour may be minimizing competition between the two.

MANAGEMENT IMPLICATIONS

Given the popularity of the site and the number of visitors each year, the potential for disturbance exists. On warm, sunny afternoons bats can often be seen in the crevice near the cliff pool as they edge their way out to cool off. Visitors occasionally peer in to see the bats, which causes them to scurry deeper into the crevice. This disturbance probably causes heat stress to the bats, although the significance of this is unknown.

Table 2. Summary of arthropods found in scats collected from bats captured during 1998. All figures are numbers of occurrences expressed as a percent of total number of occurrences for each species.

Species	Lepidoptera	Diptera	Neuroptera	Coleoptera	Hemiptera	Arachnida
M. keenii	38.5	11.5	11.5	0	3.8	34.6
M. lucifugus	36.4	27.3	27.3	4.5	0	4.5

Human activity during the evening may delay emergence. The emergence generally began about one-half hour after observed sunset, or after 2200 during May, June, and July. Current policy is that no visitors are allowed to camp on Gandl K'in, and they must leave the island by 2230, so disturbance is generally minimized.

An important part of the study was the review of redevelopment plans to identify activities that may disturb this maternity colony. A major health concern for humans is that the flow of water through the pools is insufficient to allow rapid turn over. To address this concern, it was proposed to seal the pools to prevent seepage. An investigation of the cliff pool revealed that the flow of hot water through the crevice in which the bats roost depends on the pool being full. Sealing the pool would thus alter the thermal regime of this crevice and possibly eliminate it as a roost. Plans have been altered to allow the water to continue flowing through this crevice.

ACKNOWLEDGEMENTS

I am carrying out the field work for this project under the direction of my thesis committee: Dr. R. Ring, supervisor; Dr. T. Reimchen; Dr. D. Eastman; Dr. M. Brigham; and D. Nagorsen. The following agencies have provided financial support: Gwaii Haanas National Park Reserve/Haida Heritage Site; Canadian Wildlife Service, Pacific Region; and the University of Victoria. Special thanks to R. Ring, M. Brigham, F. Burles, L. Rissling, R. LeBlanc, and a number of Park staff for field assistance; to D. Nagorsen of the Royal British Columbia Museum for sharing his records of Keen's myotis; and to T. Gore, H. Down (UVic Advanced Imaging Lab), and P. Bartier (Gwaii Haanas) for technical assistance.

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Introduced Species Management in Haida Gwaii (Queen Charlotte Islands)

Todd E. Golumbia
Gwaii Haanas National Park Reserve and Haida Heritage Site
Box 37, Queen Charlotte City, BC, V0T 1S0, Canada
todd_golumbia@pch.gc.ca

ABSTRACT

Gwaii Haanas (Gwaii Haanas National Park Reserve and Haida Heritage Site) recognizes species introductions on the archipelago of Haida Gwaii (Queen Charlotte Islands) as one of the more important threats to ecological integrity in this region. These introductions may place many unique species or natural communities at risk. Several initiatives are underway to mitigate the effects of these species as well as to reduce the risk of range expansions or new introductions. This paper provides examples that illustrate the scope of introduced species research, monitoring, and restoration initiatives that are underway on Haida Gwaii.

Key words: biodiversity, black rat, Haida Gwaii, introduced species, island, mule deer, Norway rat, protected areas, Queen Charlotte Islands, raccoon, restoration, wildlife management.

A significant threat to biological diversity, particularly in island ecosystems, is the introduction of nonnative species. Without human intervention to reverse some of the changes that have taken place, and to prevent further introductions, many unique species or natural communities may be at risk. Gwaii Haanas (Gwaii Haanas National Park Reserve and Haida Heritage Site) as part of the larger archipelago of Haida Gwaii (Queen Charlotte Islands), recognizes species introductions as one of the most important threats to ecological integrity (Golumbia 1999). While we cannot reverse the clock to when introductions were not present, a large effort has been focused on identifying those introduced species issues that are of the highest conservation priority and that are feasible, socially acceptable, and cost effective. Several initiatives are trying to mitigate the effects of these species as well as reduce the risk of range expansions or new introductions. Because of the scale and complexity of these issues, partnerships have been necessary to effectively set objectives and achieve our goals. Through the following examples including exotic mammals (rats, raccoons, and deer) and invasive plants, the scope of current introduced species research, monitoring, and restoration initiatives will be illustrated.

Like many island systems worldwide, Haida Gwaii is essentially isolated from continental influences and therefore supports a unique assemblage of plants and animals. This archipelago of several hundred islands and islets covers approximately 1 million ha in area and is located approximately 80 km offshore of northwestern British Columbia. The protected area of Gwaii Haanas (147,000 ha) encompasses the southernmost 15% of the archipelago. Other significant protected areas, including Naikoon Provincial Park and associated reserves (71,000 ha) in the northeast and V. L. Krajina Ecological Reserve (14,000 ha) on the west coast of Graham Island, represent 8.5% of the land base. Many areas outside of these disjunct reserves have been extensively logged and otherwise developed, particularly over the past 25–30 years.

The archipelago is recognized for high numbers of endemic, disjunct and rare species indicative of an insular ecosystem. In terms of species "at risk," the British Columbia (B.C.) Conservation Data Centre (1998) and the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) (1998) list 36 floral and 26 faunal species as vulnerable, threatened, or endangered. In addition, 14 plant associations are also listed. The life histories and habits of many island species reflect insular conditions where relatively few species have colonized. Exotic species tend to outcompete and hence displace indigenous forms. The effects are exacerbated in such systems because these species most often arrive without their predators or competitors.

While it is more difficult to track the introduction of flora and avifauna to the islands, the records for terrestrial mammal fauna are more certain and provide an interesting historical perspective to a currently distressing situation. Excluding humans, the terrestrial mammal fauna of Haida Gwaii consists of 13 indigenous species (one of which is extinct). Over the past 125 years, 10 mainland species have been introduced to Haida Gwaii (Table 1) resulting in almost more introduced mammal species than indigenous!

Haida Gwaii also represents some of British Columbia's

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1. Mammal species on Haida Gwaii (Queen Charlotte Islands).

Indigenous	Keen's mouse	*Peromyscus keeni keeni*
		P. keeni prevostensis
	dusky shrew	*Sorex monticolus classodon*
		Sorex monticolus prevostensis
	ermine	*Mustela erminea haidarum*
	marten	*Martes americana nesophila*
	river otter	*Lontra canadensis periclyzomae*
	black bear	*Ursus americana carlottae*
	Dawson caribou	*Rangifer tarandus dawsoni* (extinct)
	Keen's long-eared myotis	*Myotis keenii*
	little brown bat	*Myotis lucifugus alascensis*
	California myotis	*Myotis californicus caurinus*
	silver-haired bat	*Lasionycteris noctivagans*
Introduced	house mouse	*Mus musculus*
	black rat	*Rattus rattus*
	Norway rat	*Rattus norvegicus*
	red squirrel	*Tamiasciurus hudsonicus lanuginosus*
	raccoon	*Procyon lotor vancouverensis*
	beaver	*Castor canadensis leucodontus*
	muskrat	*Ondatra zibethicus osoyoosensis*
	Sitka mule deer	*Odocoileus hemionus sitkensis*
	European red deer	*Cervus elaphus elaphus*
	Elk	*Cervus elaphus nelsoni*

most extensive seabird nesting habitat, with internationally significant populations of some species. Of the 1.5 million seabirds breeding at numerous sites in Haida Gwaii, almost all nest in or on the ground with little or no natural defence to predation. The introduction of efficient terrestrial predators has caused severe losses on even the largest colonies. Several seabird species have been assigned a risk status based on the threats posed by introduced predators.

RATS

Rats have a long association with humankind. They are revered in some cultures but despised in most. A Jain temple located at Deshnoke in the state of Rajasthan is a place where the rat is sacred and thousands reside in the small shrine. In contrast, the province of Alberta proudly claims "rat-free" status and boasts that "every rat that ends up in the province, ends up dead" (Hodgson 1997). Rats are highly successful colonizers and have spread across the globe as stowaways on many early maritime explorations. The black rat (*Rattus rattus*) has probably been on Haida Gwaii since the late 1700s and may have been here earlier. More recent shipping activity has introduced the larger more aggressive Norway rat (*Rattus norvegicus)* to many islands in Haida Gwaii. Rats are voracious predators, and on those islands where they have been introduced, many seabird species have disappeared or are in decline (Harfenist and Kaiser 1997). Eighteen islands in the archipelago are known to support rats (Bertram and Nagorsen 1995).

Langara Island, located at the northern extreme of Haida Gwaii, was historically one of British Columbia's largest seabird colonies. However, breeding populations of seabirds have either been lost or have declined precipitously. For example, the ancient murrelet, a COSEWIC-listed species, has declined from an estimated historic population of 200,000 birds to a 1988 estimate of 24,000 (Bertram 1995). This decline has partly been due to rats (Kaiser et al. 1997). Several other islands have exhibited a similar trend in the presence of rats.

Restoration techniques developed and used successfully for rat control in New Zealand over the past 10 years have been used to clear rats from seabird colonies and thus enable seabird populations to recover. The approach uses toxic baits at fixed stations (Kaiser et al. 1997). The key to this program's success is the regular and continuous supply of baits within every possible rat home range. The bait is a second-generation coumarin-based anti-coagulant which is effective after 1 feeding; more traditional first-generation compounds require multiple feedings and are therefore more susceptible to feeding avoidance behaviour and resistance by the target organism (Kaiser et al. 1997).

The eradication project initiated by the Canadian Wildlife Service (CWS) on Langara, Lucy, and Cox islands has been successful. At the other end of the archipelago, Gwaii Haanas has successfully removed rats from St. James Island, which was also a large seabird colony island before the invasion of

rats. Based on the expertise of Gary Kaiser (CWS) and Rowley Taylor (R.H. Taylor Associates), Gwaii Haanas staff improved the methodology and bait station design to further prevent nontarget poisoning in the St. James project. Ten islands within Gwaii Haanas still have rats on them. The intent is to continue with this restoration program on those islands where this type of program is feasible. Ecosystem recovery will also be monitored.

In addition to active restoration, long-term success will require that people do not inadvertently reintroduce rats to restored islands or to any new islands. Gwaii Haanas staff have developed a communications program to inform people about the risks of transporting rats from infected sites to clean sites. The program includes media information releases, backcountry orientation programs, and an information pamphlet which is distributed widely.

RACCOONS

The raccoon (*Procyon lotor*) was introduced to Graham Island in the 1940s to augment the fur trade. This highly adaptive and opportunistic mammal has successfully colonized many islands in the archipelago. Several surveys in the late 1980s and early 1990s determined the range of the raccoon. Research indicated that raccoons opportunistically swim as far as 1 km to remote islands (Hartman 1993). This finding places over half of the burrow-nesting seabirds on Haida Gwaii at risk (Working Group On Raccoon-Seabird Interactions 1995*a*). In response to this threat, a working group was formed in 1994 with representatives from the British Columbia (B.C.) Ministry of Environment, Lands and Parks; CWS; the Archipelago Management Board (Parks Canada and Council of the Haida Nations); and the Laskeek Bay Conservation Society. This group developed a monitoring and control protocol (Working Group On Raccoon-Seabird Interactions 1995*b*). Overall, monitoring and control programs appear to be successful. However, the protocols are being revised to reflect increased knowledge and field experience .

While the raccoon–seabird interaction was considered the most significant conservation issue, the raccoon may also threaten nesting shorebirds, other ground nesting birds, and small mammals, as well as intertidal organisms. Gwaii Haanas plans to expand our efforts to these areas, particularly looking at the long-term effect of raccoon population reduction on the larger islands and in key ecological areas (Golumbia 1999).

DEER

The Sitka mule deer has a relatively short history of just over a century on the islands. These deer were introduced by the B.C. Game Commission at least 5 times between 1880 and 1925 (Carl and Guiguet 1972, Pojar et al. 1980). Even though few individuals were introduced, they have flourished without competition or predation. Hunting statistics over a 20-year period (1976–1996) indicate that >34,000 animals have been harvested, in addition to food gathering by the Haida Nation (Laskeek Bay Conservation Society 1998). While the impact from browsing deer varies across the landscape, this species has spread to virtually all of the islands in the archipelago with only 8 islands known to be deer-free. These sites are small offshore islands representing approximately 500 ha which is <0.1% of the archipelago's land base.

The large-scale impact of deer on the forest ecology of Gwaii Haanas is obvious and has been recognized since the late 1970s (Pojar et al.1980). Work in the Gwaii Haanas area by Jean Louis Martin and others in the early 1990s began to measure the effects of deer across the landscape (Martin et al. 1995). Because of this initial work, we began to realize the complexity of the problem and the potential costs associated with addressing the issues. To address these problems, a multidisciplinary research collective was formed in 1995 with interested parties including the Laskeek Bay Conservation Society, Gwaii Haanas Archipelago Management Board, CWS, the Centre National de la Recherche Scientifique (France), and the B.C. Ministry of Environment, Lands and Parks. This group, now known as the Research Group on Introduced Species (RGIS), has initiated a 5-year research program funded by Forest Renewal British Columbia (FRBC), the South Moresby Forest Replacement Account (SMFRA), and the participating agencies. The purpose of this research is to better understand the interaction of introduced species (particularly deer) on the forest ecology of Haida Gwaii. The network of principal investigators and research collaborators has enabled us to investigate several hypotheses simultaneously, the results of which will be useful in designing management solutions in the future.

RGIS is now into its second year of work and much has been accomplished toward achieving the following research objectives: (1) to understand the interactions among flora and fauna shaping forest ecology, forest economy, and biodiversity conservation; (2) to predict vegetation recovery potential if Sitka mule deer are controlled (and the consequences of leaving them uncontrolled); (3) to understand spatial and temporal movements of Sitka mule deer among islands and between habitats; and (4) to understand the individual and synergistic effects of Sitka mule deer and red squirrel on the ecology of forest birds.

Deer browsing has reduced abundance and vigour of virtually all species of shrubs and herbs; in extreme cases, the understory structure of the forest is absent (Daufresne and Martin 1997). The direct effects of browsing are obvious and can range from reduced vigour to elimination of certain species. The indirect effects on forest structure and function

are less obvious but equally important. These plants and their fruits are no longer available for species that use them for food. The physical structure from the forest floor to the canopy in a natural forest provides important habitat for nesting, foraging, and cover used by birds and small mammals. This structure is greatly reduced and simplified in Haida Gwaii forests (Martin et al. 1995, Daufresne and Martin 1997). More subtle effects are likely occurring at the forest floor with regard to nutrient cycling, terrestrial arthropods, and soil structure.

Preliminary research suggested that this loss of cover increases the vulnerability of nesting song birds to predation by another introduced species, the red squirrel (Martin et al. 1996). However, the more significant effect is apparently an increase in activity by native predators who benefit from the increased visibility in the forest (J. L. Martin, Centre National de la Recherche Scientifique, Montpellier, France, 1998, pers. comm.). The lack of understory is also reflected in the growth and survival of tree seedlings and subsequent recruitment into the forest canopy. Heavy pressure on western redcedar (*Thuja plicata*) seedlings, which are a preferred browse species, also threatens the natural regeneration of that species (Baltzinger and Martin 1998). This is of particular concern to forest companies who must regenerate redcedar after harvesting on cedar sites, and to the Haida people, who consider redcedar and other forest plants as central to their culture.

Deer densities are much reduced on the mainland, where natural predators and hunters control numbers. Liberal hunting quotas have been in place throughout Haida Gwaii and are still used outside Gwaii Haanas. Vegetation plots across the islands indicate that hunter accessible areas have up to 3 times more redcedar survival to the sapling stage than more isolated areas (Baltzinger and Martin 1998). However, the effect on deer numbers is localized.

Eradication of deer on Haida Gwaii is neither feasible nor socially desirable. However, deer could be managed on islands sufficiently distant from a "source" population to make eradication or continued control feasible. This proposal would provide "deer-free" islands to represent a benchmark area. To test this concept, an experimental deer cull has been undertaken on 2 islands. Baseline measurements of forest vegetation as well as song bird numbers and diversity have been established on the islands and on adjacent control areas. The results are visible and show obvious reestablishment of the herb and shrub layer after only 1 growing season.

To develop management scenarios for the island deer population, known models for the Sitka mule deer which have been developed elsewhere on the coast must be tested. Radiotelemetry, in combination with DNA analysis, will be used to better understand inter-island movement of deer as well as seasonal movements and habitat use on large and small islands. This information will be useful in determining where additional control efforts may be successful.

EXOTIC VEGETATION

There has been a long standing interest in the native flora of Haida Gwaii. Numerous records exist of rare species as well as species that are endemic to these islands or which exhibit disjunct distributions indicative of glacial refugia (Ogilvie 1989). Native vascular flora on the islands totals 657 species of which 34 are considered rare and 10 are considered local or regional endemics. Some 143 species (22%) are considered introduced to the islands, several of which are considered invasive (Lomer and Douglas in press).

Gwaii Haanas staff have begun to document historic and current distribution of introduced plant species. Most exotic plant introductions in Gwaii Haanas appear to be localized around human disturbed sites such as historic logging camps and roads, homesteads, village sites, mining sites, and lighthouses (Fellows 1996). The more invasive species are a particular concern because they spread easily and can outcompete native species especially where disturbance has occurred. The spread of high risk species is also being monitored. Where introduced species are limited in distribution and accessible, attempts will be made to remove the plants before they spread.

The agricultural and horticultural traditions in our cultures continue to support the presence of exotics in the regional ecosystem. Current logging practices, including follow-up activities such as slope stabilization and road deactivation, can introduce invasive weed species through hydro-seeding or other operations. Recent landscape practices have used species such as Scotch broom (*Cytisus scoparius*) and gorse (*Ulex europeus*). These species are widespread invasive plants in the south but are still limited in their distribution on Haida Gwaii and are not yet found within Gwaii Haanas. In all cases, public education and awareness are the best methods of prevention.

DISCUSSION

It is difficult, likely impossible, to fully understand the impact of introducing species into a complex array of native plants and animals. However, we know that introducing nonnative species anywhere in a natural system can place native species and natural ecosystems at risk. Reversing the changes are difficult, costly, and sometimes impossible. Societal and cultural values must also be considered and can often be equally complex. What is our role in restoring ecosystems? Is there more value in eradicating a "vermin" such as the rat or the raccoon than a species of higher cultural value such as the deer?

The complex nature of these issues confirms the need to adopt an adaptive management approach which combines

research with monitoring and active restoration. The spatial scale of these issues, from island complexes to global distribution, demands that partnerships be forged to develop strategies for reducing the effects of these introductions. Through the research collective that has been established (RGIS), an international network of experts combined with local people and local knowledge can work together to address these issues and collect the information needed to make sound management decisions.

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Monitoring and Control of Raccoons on Seabird Colonies in Haida Gwaii (Queen Charlotte Islands)

Anne Harfenist
Canadian Wildlife Service
R.R.1, 5421 Robertson Road, Delta, BC, V4K 3N2, Canada
harfenis@mail.bulkley.net

Karen R. MacDowell
British Columbia Parks
101-3220 Eby Street, Terrace, BC, V8G 5K8, Canada

Todd Golumbia
Parks Canada, Box 37
Queen Charlotte City, BC, V0T 1S0, Canada

George Schultze
British Columbia Ministry of
Environment, Lands and Parks
Bag 5000, Smithers, BC, V0J 2N0, Canada

Laskeek Bay Conservation Society
Box 867
Queen Charlotte City, BC, V0T 1S0, Canada

ABSTRACT

More than 1.5 million seabirds breed in Haida Gwaii, B.C. Most are burrow-nesters concentrated at 64 colony sites. These globally significant seabird colonies have been put at risk due to the introduction of raccoons (*Procyon lotor*) to enhance the local fur trade in the 1940s. Evidence collected in the early 1990s indicated that raccoons pose a serious threat to approximately one-half of the burrow-nesting seabirds that breed in Haida Gwaii. Significant proportions of the world population of ancient murrelets (*Synthliboramphus antiquus*) are at risk. In response to this threat Parks Canada, Environment Canada's Canadian Wildlife Service, the Laskeek Bay Conservation Society, and the British Columbia Ministry of Environment, Lands and Parks (B.C. Environment, Wildlife Branch, and B.C. Parks) formed an interagency working group to develop a monitoring and control program. Three factors influenced the selection of seabird colony islands for annual monitoring: the number and species of seabirds present; island accessibility to raccoons; and the presence of other introduced predators. The islands chosen support approximately 90% of all ancient murrelets, rhinoceros auklets (*Cerorhinca monocerata*), and storm petrels (*Oceanodroma* spp.), and about 70% of Cassin's auklets (*Ptychoramphus aleuticus*) breeding in Haida Gwaii. All raccoons observed on seabird colony islands during the annual surveys were killed. In addition, raccoons have been killed at 1 off-colony site that had consistently served as a source area. 1998 was the first year in which no raccoons were found on any of the monitored seabird islands. Analysis of the results from the first 5 years of monitoring and control has allowed the working group to modify the program. The revised program will be less costly and labour-intensive, while still providing protection for the nesting seabirds.

Key words: ancient murrelet, auklet, *Cerorhinca monocerata*, Haida Gwaii, introduced species, predation, *Procyon lotor*, *Ptychoramphus aleuticus*, Queen Charlotte Islands, raccoon, seabird, *Synthliboramphus antiquus*.

Introduced predators are considered to be among the greatest threats to seabird conservation worldwide (e.g., Johnson and Stattersfield 1990). Widespread declines and extirpations in seabird populations within the past century have been attributed to predation by nonnative mammals (Moors and Atkinson 1984). Raccoons (*Procyon lotor*) were deliberately introduced to Graham Island in Haida Gwaii (Queen Charlotte Islands; Fig. 1) by the Provincial Game Commission to enhance the local fur trade in the early 1940s (Carl and Guiget 1972). By 1992, raccoons had spread through the archipelago and had reached some of the smaller islands that support breeding colonies of seabirds. Seabirds possess few morphological or behavioural defenses against mammalian predators. Furthermore, their habit of nesting in colonies during well-defined breeding seasons leave the birds concentrated in time and space. Those habits increase the probability that a terrestrial predator that reaches a colony island will have an impact on a significant portion of the birds' populations.

By the late 1980s, biologists had collected circumstantial evidence that raccoons were negatively affecting some seabird colonies in Haida Gwaii (e.g., Rodway 1991). Those observations prompted the formation of the Working Group on Raccoon-Seabird Interactions (referred to as "the Working Group" for the purposes of this paper), made up of representatives of the British Columbia Ministry of Environment, Lands and Parks (B.C. Parks and B.C. Environment),

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Environment Canada (Canadian Wildlife Service), the Laskeek Bay Conservation Society, and Parks Canada. The Working Group assessed the evidence for raccoon impacts on seabirds and developed a management plan.

This paper summarizes the management plan and presents results from the first 4 years of implementation. Also discussed are proposed modifications to the management plan that will result in a less costly and labour-intensive program, and yet still provide protection for the nesting seabirds of Haida Gwaii.

BACKGROUND

More than 1.5 million seabirds breed at 180 sites in Haida Gwaii (Rodway 1991). Nine of the 12 seabird species that breed on the archipelago are burrow- or ground-nesters and, thus, vulnerable to mammalian predators. The burrow-nesting species represent approximately 99% of the total regional population of seabirds and are concentrated in colonies at 65 sites. Those sites support an estimated 50, 16, and 2% of the world's populations of ancient murrelets (*Synthliboramphus antiquus*), Cassin's auklets (*Ptychoramphus aleuticus*), and rhinoceros auklets (*Cerorhinca monocerata*), respectively (Gaston and Masselink 1997). The latter 2 species are provincially Blue-listed, and the ancient murrelet has been designated as Vulnerable by COSEWIC, due to the threat posed by introduced predators (Gaston 1994). Tufted puffins (*Lunda cirrhata*) are also Blue-listed.

Figure 1. Haida Gwaii, showing the place names used in the text.

Another species of special concern is Peale's peregrine falcon (*Falco peregrinus pealei*). Approximately two-thirds of the British Columbia population of Peale's peregrine falcons, Blue-listed and designated as Rare by COSEWIC, nest in Haida Gwaii (Campbell et al. 1990, Edie 1995). These raptors depend heavily on ancient murrelets as a food source (Nelson 1977) and have declined on at least 1 site where introduced predators eliminated a large proportion of the seabirds (Gaston 1994).

In addition to being an integral component of the Haida Gwaii ecosystem, seabirds of this region represent an important part of the cultural background of the Haida. They are also valued by tourists and, thus, are of economic benefit to the people of Haida Gwaii.

Circumstantial evidence of seabird colony destruction by raccoons in Haida Gwaii has been noted over many years: of 7 known cases of colony abandonment by breeding seabirds, 5 occurred on islands where raccoons have been documented (e.g., Rodway et al. 1988, Gaston 1994). In the early 1990s, 2 studies provided direct evidence of the tremendous destructive potential of raccoons to breeding seabirds (Gaston and Masselink 1997, Hartman et al. 1997).

In 1991, 3 raccoons on East Limestone Island killed an estimated 6% of the approximately 1,200 pairs of breeding ancient murrelets and caused reproductive failure in approximately 10% of the pairs (Gaston and Lawrence 1993, Hartman et al. 1997). Raccoons excavated burrows and ate eggs, chicks, and adults. They also exhibited surplus killing tendencies. The mammals were eradicated prior to the 1992 breeding season. In 1992, predation rates decreased by about 80% from the levels recorded in 1990 (n = 82) and 1991 (n = 106), and no burrow excavations, broken eggs, dead chicks, or headless carcasses were observed (Gaston and Lawrence 1993, Hartman et al. 1997).

On Helgesen Island, where 9–12 raccoons were sighted in 1993, approximately 30,000 burrow-nesting seabirds were eliminated between 1986 and 1993 (Gaston and Masselink 1997). Numbers of rhinoceros auklets, ancient murrelets, and Cassin's auklets on the island declined by approximately 79, 83, and 95%, respectively. The 10,000 pairs of rhinoceros auklets that were lost represented about 30% of the Haida Gwaii breeding population. Evidence that raccoons were responsible for the declines included a large number of burrows that had been excavated by the mammals. In addition, all fresh raccoon scats found on active colony areas contained seabird feathers. Over the same 7-year period, seabird populations on nearby raccoon-free islands increased or were stable (Gaston and Masselink 1997).

By 1992, raccoons had spread to 86% of all islands surveyed in the archipelago that were within 400 m of a

Table 1. Approximate percentage of Haida Gwaii breeding populations of burrow-nesting seabirds covered by 1995 and 1999 management plans. Calculations do not include islands with introduced rats.

Year of plan	Percentage of population covered by plan				
	ANMU[a]	CAAU	RHAU	TUPU	Petrel spp.
1995	87	71	89	5	91
1999	82	89	89	82	85

[a] ANMU = ancient murrelet; CAAU = Cassin's auklet; RHAU = rhinoceros auklet; TUPU = tufted puffin; petrel spp. = *Oceanodroma* spp.

potential source site for dispersing raccoons, and to 1 island approximately 1,000 m from a potential source area (Hartman 1993). Over half of the burrow-nesting seabirds in Haida Gwaii breed on islands within 1,000 m of a potential source area. Although the accessibility of an island to raccoons is influenced by factors other than over-water crossing distance, such as exposure and water currents, the figures above strongly suggest that a large proportion of the archipelago's seabirds are potentially at risk. To date, raccoons have reached at least 7 and possibly 11 islands that support extant seabird colonies (Harfenist and Kaiser 1997).

1995 MANAGEMENT PLAN

In response to the threat posed by raccoons to seabird populations on Haida Gwaii, the Working Group developed a management plan (Working Group 1995*a*). The plan consists of 4 main components: monitoring, control, research, and communication. Several agencies are involved in the issue of raccoon-seabird interactions: the Archipelago Management Board (Council of Haida Nations and Parks Canada); British Columbia Ministry of Environment, Lands and Parks (B.C. Environment, Wildlife Branch, and B.C. Parks), and Environment Canada (Canadian Wildlife Service). Each of the organizations supports the concept of control of introduced species as a means of protecting native species, therefore mandates are consistent with regard to this type of resource management issue.

Monitoring

Our current understanding of raccoon distribution is based primarily on observations of the animals and/or their scats from the last 15 years (e.g., Rodway et al. 1988, van den Brink 1992) and the results of the first 5 years of a raccoon monitoring program (Working Group unpubl. data), and provides limited insight into the raccoon's long-term patterns of range expansion. In the absence of the ability to predict dispersal patterns and probabilities, the Working Group endorsed a large-scale, ongoing, annual monitoring program.

The management plan lists islands that should be checked annually for the presence of raccoons and denotes which government agency is responsible for each island. The responsibility for annual monitoring lies with Parks Canada within Gwaii Haanas, with B.C. Parks within ecological reserves, and with the Canadian Wildlife Service in all other areas. The plan refers to a supplementary monitoring plan (Working Group 1995*b*) that presents the rationale for including or excluding islands. The islands were chosen on the basis of 3 main criteria: probability of access by raccoons (based on over-water crossing distance, exposure, and water currents); size and composition of the seabird population; and status of the island with regard to rats, another introduced predator. In general, large nearshore colonies are of higher priority than are smaller, more offshore ones. Colonies with rats are not included in the annual monitoring list. The selected islands support approximately 90% of all ancient murrelets, rhinoceros auklets, and storm petrels (*Oceanodroma* spp.), and 70% of all Cassin's auklets breeding in the archipelago (Table 1).

Non-colony islands that may be important as stepping stones for raccoons between source areas and colony islands have also been identified (Working Group 1995*b*). The decision about the frequency at which such islands should be monitored was left up to the responsible agency.

Two main methods are outlined for detecting raccoons: scat searches, and spotlight surveys. The former method involves intensive searches for raccoon scats and latrines along a standardized shoreline transect. The transects are approximately 10 m wide and extend from the vegetation edge into the forest. Spotlight surveys involve skirting part or all of an island perimeter by boat at night, while scanning the shoreline and intertidal area with a 750,000+ candlepower spotlight. Raccoons can be detected by the reflection from their eyes. Details of the above methods are given in the monitoring plan (Working Group 1995*b*). Searching for evidence of predation has only limited usefulness as a detection method, because most carcasses are consumed or converted to feather piles within 24 hours (Hartman et al. 1997). The reliability of the monitoring methods is unknown.

Control

The management plan recommends that a control program be carried out on all colonies reached by raccoons. Given the high level of destruction that a small number of raccoons is capable of, the goal of control must be the complete eradication of the mammals from seabird colonies and should be

carried out in the same year as the raccoons are detected. The responsibility for carrying out control of raccoons on seabird colonies in Haida Gwaii rests with Parks Canada within Gwaii Haanas, with B.C. Parks within ecological reserves, and with the Wildlife Branch in all other areas.

The principle method used to control raccoons is hunting at night from a small boat. A spotlight is used to spot the raccoon; the animals are killed using a shotgun (see Working Group 1999 for details). Only skilled and trained staff are permitted to participate in the control, in order to minimize the probability that animals are wounded rather than killed humanely. Hunting with dogs and trapping have been used to control raccoons elsewhere in North America, but an attempt to use dogs on Limestone Island in 1992 failed because the dogs were unable to follow the raccoons over the steep and rocky terrain (Hartman 1994); thus, the method was not recommended for use on most seabird islands.

Removal of raccoons from shorelines adjacent to colony islands in order to reduce the probability of migration is considered in the management plan. However, the Working Group recommended that such a strategy should not be implemented unless a research project is conducted that demonstrates the effectiveness of the technique.

Research

Early research conducted on the interactions between raccoons and seabirds significantly increased our knowledge of the raccoon's current and potential impact on breeding seabirds of Haida Gwaii and formed the basis of the present management plan (Gaston and Masselink 1997, Hartman et al. 1997). However, because raccoons can cause severe damage within a single seabird breeding season, methods of detection and removal must be effective in the early stages of colonization, when raccoon densities may be low. Thus, the research priority is to evaluate the effectiveness of different monitoring and control methods under the environmental conditions found in Haida Gwaii. This includes an evaluation of preventative raccoon control from possible source areas to reduce the rate or probability of dispersal.

Communication

The management of raccoon-seabird interactions crosses the jurisdictional boundaries of several government agencies and is also of concern to non-government organizations and individuals. Annual meetings of the Working Group facilitate information exchange. As part of the plan, Parks Canada developed a central database for the results of all monitoring and control efforts, which ensures that information is not lost with changing personnel. In acknowledgment of increasing public involvement in conservation issues and the sensitivity of control programs, the plan encourages members of the Working Group to ensure that the public is kept informed about the issue and surrounding activities.

RESULTS

Monitoring

In the 4 years since the implementation of the management plan, the 31 seabird colony islands identified as important and vulnerable have been intensively monitored for the presence of raccoons. The results of these surveys indicated that raccoons are no longer on several of the islands from which they had previously been reported: no new signs of raccoons have been noted on Instructor, Rock, Skincuttle, George, Ramsay, or Skedans islands (Working Group unpubl. data). In some cases, the original reports may have been incorrect; for example, raven castings may have been misidentified as raccoon scats (Working Group 1995*b*). For the other islands, the results suggest that either the raccoon(s) left the island or that the individual(s) died before establishing a persistent population. Evidence of new populations of raccoons have been found 3 times during the annual surveys: on Alder and Helgesen islands, and the Limestone Islands (Gaston and Heise 1994, Working Group unpubl. data). In the case of Alder Island, a raccoon latrine was found, but the lack of evidence of raccoons during subsequent spotlight surveys and searches for signs of predation on the colony suggests that the raccoon either returned to nearby Burnaby Island or died.

The low between-island dispersal rate of raccoons indicated by the above results has given the Working Group the confidence to remove 8 of the 31 islands from the list of sites to be checked on an annual basis (Table 2; Working Group 1999). The majority of the islands removed from the list are >1,000 m from a potential source area for raccoons. This reduced effort will allow agencies to focus more on some of the small and easily monitored non-colony islands that are potential stepping stones to important colonies. For example, Lucy Island has been added to the list in light of the successful eradication of rats from the Langara group of islands (Kaiser et al. 1997). Although Lucy Island no longer supports breeding seabirds, it is a stepping stone to Langara and Cox islands, which still support significant populations of ancient murrelets, tufted puffins, and peregrine falcons.

By collaborating, the responsible agencies have been able to decrease individual efforts in terms of human and fiscal resources, yet maintain an effective management program. For example, a Canadian Wildlife Service research crew conducted the monitoring on Frederick Island for B.C. Parks from 1996 to 1998. B.C. Parks also established a partnership with the Haida Fisheries Program, which now monitors Lepas Islet. Cost savings are estimated to have been $3–5,000/year for each agency.

Control

By 1998, the Wildlife Branch had eradicated raccoons at 2 sites: Limestone Islands (Schultze 1994), and Helgesen Island (Gaston and Masselink 1997, Harfenist unpubl. data).

Table 2. Islands selected for annual monitoring for raccoons (Working Group 1995a). The 8 islands listed in bold have been dropped from the revised monitoring scheme; the 6 islands in italics have been added (Working Group 1999).

Island	Population of breeding burrow-nesting seabirds[a] (no. of pairs)					Distance to source (m)	Comments
	ANMU	CAAU	RHAU	TUPU	Petrel spp.		
Cape Kuper	10	100	10			<50	
Rogers	1,700	40	20		28,700	100–200	
Lepas		200			8,000	"	
Instructor[b]					1,600	"	
Helgesen[b]	1,100	200	3,300		200	200–300	
Willie	10	200	80			"	
Alder[b]	14,000	3,200			60	"	
Rainy					100	"	
Limestones[b]	1,500	40				300–400	
Luxmoore	1,000	400	300		5,700	"	
Carswell	1,700	200	20		300	400–500	
Shuttle						"	stepping stone
Bischof					50	500–600	stepping stone
Skung Gwaii	200	24,700	13,600	16	10,700	"	
Charles			100		100	600–700	
Annette			20			"	stepping stone
Hippa	40,100	12,500		20	23,700	"	
Rock[b]		5,100			17,000	"	
Skincuttle[b]	2,200	1,000			4,300	"	
George[b]	11,600	5,900				"	
Jeffrey	1,000	2,700				"	
East Copper	4,400	10,900			40	"	
Frederick	68,400	89,900			100	700–800	
Faraday						"	stepping stone
Lyell Bay						"	stepping stone
Bolkus	9,900	1,000	20		200	800–900	
Gordon		600	100			"	
Langtry					12,300	"	
Ellen						900–1,000	stepping stone
Hotspring	6	10			900	>1,000	
Lihou	12,100	13,100	2,800	13	13,500	"	
Rankine	26,000	26,000			14,300	"	
Lucy						"	stepping stone
Howay	300	300			10	"	
Skedans[b]		100			1,100	"	
Ramsay[b]	18,200	12,900				"	
House	2,600	40				"	

[a] Population estimates from Rodway (1991), Gaston and Masselink (1997), and Harfenist (1994); estimates >100 have been rounded to nearest 100. ANMU = ancient murrelet; CAAU = Cassin's auklet; RHAU = rhinoceros auklet; TUPU = tufted puffin; petrel spp. = *Oceanodroma* spp.

[b] These islands have been reached by raccoons (Hartman 1993); evidence for George, Rock, and Instructor is questionable.

Three raccoons were removed from East Limestone Island prior to the 1992 seabird breeding season. The control of raccoons at that site reduced the predation rate in the seabird colony by 80% (Hartman et al. 1997). On Helgesen Island, 3 years of control were required to remove all the raccoons: a total of 7 raccoons were killed (Gaston 1993, Harfenist unpubl. data). A single raccoon, presumed to be a new immigrant, was sighted on the island in 1997 and was eliminated (Harfenist unpubl. data). The control efforts on Helgesen have been successful: ancient murrelets, Cassin's auklets, and rhinoceros auklets have begun to recolonize areas from which they had been almost completely eliminated (Harfenist unpubl. data); 1998 was the first raccoon-free year on Helgesen Island in at least 6 years.

The success of the control efforts at both Limestone and Helgesen islands was strongly influenced by season, local

topography, and weather. The experience gained during nights of hunting raccoons have allowed the Working Group to modify the control protocols to increase the chances of success (Working Group 1999). For example, control efforts on seabird colonies are most effective if conducted outside the seabirds' breeding season, when raccoons are highly visible as they forage down in the intertidal zone. Predators are difficult to see and shoot when they are feeding in the uneven and elevated terrain of the seabird nesting habitat.

Control actions have only been initiated when raccoons have been identified on a seabird colony island. Though some control has been done on adjacent shorelines at the same time, a systematic control program on adjacent shorelines has not been adopted. Although it is likely that removal of raccoons would reduce the probability of raccoons migrating across to colony islands, there is still uncertainty as to the potential effects that removal will have on dispersal behaviour. Crews from the Laskeek Bay Conservation Society and the Wildlife Branch have periodically shot raccoons along the shoreline from which the mammals migrate over to the Limestone Islands. Thirty raccoons have been removed since 1991 (G. Schultze unpubl. data). No new raccoon colonizations have occurred on the Limestones since the 1994 seabird nesting season. Those limited results suggest that further investigation of this technique should be carried out.

Research

Research has been aimed at improving monitoring and control techniques. The Habitat Conservation Fund provided support to the University of Northern British Columbia to investigate the relative effectiveness of various methods. The research program investigated the efficacy of using infrared imaging from the air as a tool for detecting raccoons (Gilbert 1996). The conclusion was that the infrared system is effective at detecting raccoons foraging during low tide when ambient conditions are cool. However, tests were only conducted in areas of high raccoon density and lacked controls. The research also tested hunting raccoons with trained dogs and trapping as control methods. The former method has been successful only in situations where the terrain is flat and free of obstacles (e.g., on sandy beaches or in large estuaries), while on many islands in the archipelago raccoons can traverse topography where dogs and/or humans cannot safely follow. Trapping is labour-intensive and time-consuming. The easy availability of alternative high-quality food sources makes baiting and attracting the animals difficult.

Communication

Communication between the organizations involved in the management of raccoon-seabird issues has been critical to the development and updating of standardized methodologies. A formal annual meeting has now been scaled back to periodic conference calls. Members of the Working Group have participated in numerous public information sessions including talks, radio and newspaper interviews, and displays. The Laskeek Bay Conservation Society and Gwaii Haanas have featured introduced species in their newsletters, which are distributed to a wide audience.

CONCLUSIONS

After 4 years of implementation, the Working Group is confident that its activities are having a significant positive effect on the conservation of seabirds and their habitats at both local and regional scales. Continued monitoring is required to ensure that extant seabirds are not put further at risk from raccoons. However, analysis of the results to date suggests that a revised program that is less costly and less labour-intensive can still provide protection for the nesting seabirds.

The management plan described here was developed to limit the effects of raccoons on breeding seabirds. However, other parts of the intertidal, shoreline, and forest communities are also at risk from raccoons. These concerns remain to be addressed.

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Impacts and Management of the Alien Eastern Gray Squirrel in Great Britain and Italy: Lessons for British Columbia

Corrie Bruemmer[1]
NPI Red Alert NW, Cumbria Wildlife Trust,
Brockhole, Windermere, Cumbria LA23 1LJ, United Kingdom

Peter Lurz
Centre for Land Use and Water Resources
Porter Building, Newcastle University
Newcastle Upon Tyne
NE1 7RU, United Kingdom

Karl Larsen
Department of Forestry
and Natural Resource Sciences
University College of the Cariboo
P.O. Box 3010
Kamloops, BC, V2C 5N3, Canada

John Gurnell
School of Biological Sciences
Queen Mary and Westfield College
University of London
E1 4NS, United Kingdom

ABSTRACT

Introduced species are a most serious threat to biodiversity, second only to habitat degradation. The eastern gray squirrel (*Sciurus carolinensis*) is native to eastern North America, but has been introduced to several countries outside its natural range. In Europe, it has been introduced to Britain, Ireland, and Italy, and in all 3 countries it has spread and replaced the native European red squirrel (*S. vulgaris*). There are several possible explanations for the replacement of the European red squirrel by eastern gray squirrels, but the main hypotheses involve exploitation competition between the 2 species, although a disease that is fatal to red squirrels and possibly spread by the gray squirrel also may have contributed. Gray squirrels cause economic damage to forests by removing bark from trees, particularly broadleafed types such as oak and beech, and have the potential to suppress natural forest regeneration. In North America, eastern gray squirrels have been introduced into many areas in the Pacific Northwest, and throughout the southern part of Vancouver Island. They appear to be increasing their range and population densities. This paper outlines the history of eastern gray squirrel introductions to Britain and Italy, the effects on native species, and the management options currently in use and under evaluation. From this knowledge, we predict that gray squirrels on Vancouver Island may have detrimental impacts on the native North American red squirrel (*Tamiasciurus hudsonicus*) and the endangered Garry oak (*Quercus garryana*) ecosystem.

Key words: competition, eastern gray squirrel, Garry oak, Great Britain, introduced species, Italy, *Quercus garryana*, red squirrel, *Sciurus carolinensis*, *Sciurus vulgaris*, *Tamiasciurus hudsonicus*.

Introduced species are one of the most serious threats to biodiversity, second only to habitat degradation (Clout and Lowe 1996, Crooks and Soule 1996, Illueca 1996). On a global scale, almost 20% of all the vertebrates in danger of extinction are threatened in some way by alien species (Berntsen 1996). Introduced species cause dramatic changes to their new ecosystems. They physically alter the habitat, threaten predator-prey relationships, out-compete native species for resources, and act as vectors for disease (Lever 1994).

The eastern gray squirrel (*Sciurus carolinensis*), hereafter called simply the gray squirrel, is just 1 example of a species introduced to areas outside its natural range. Its natural range extends from the Gulf of Mexico, north throughout the eastern United States to southern Manitoba in the west, and to the Canadian Maritimes, southern Quebec, and southern Ontario in the east (Hall and Kelson 1959). It is adapted to hardwoods and is found in oak (*Quercus* spp.), oak–hickory (*Quercus* spp.–*Carya* spp.), and beech–maple (*Fagus* spp.–*Acer* spp.) forests (Gurnell 1987, Koprowski 1994). However, it is highly adaptable and will thrive in urban environments as well as in parks and gardens, and is known to have expanded into mixed-conifer forests. These

1 Present address: Haverbrack House Cottage, Haverbrack, Milnthorpe, Cumbria, LA7 7AH, UK
113105.1533@compuserve.com

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

characteristics make the gray squirrel a serious threat to other squirrel species such as the European red squirrel (*S. vulgaris*), the western gray squirrel (*S. griseus*), and possibly the North American red squirrel (*Tamiasciurus hudsonicus*), once introduced into their habitats.

Gray squirrels have been introduced to various locations around the world, including parts of Canada, Europe, South Africa, and Australia (Davis 1950, Corbet 1978, Seebeck 1984, Gurnell 1987). In Europe they were introduced to Britain and Ireland, as well as Italy (Middleton 1930, Shorten 1954, Currado et al. 1987), and have caused serious economic and ecological damage (Gurnell 1996*a*). In Britain, gray squirrels are considered a serious forest pest, damaging timber through their bark-stripping behaviour, and are a threat to European red squirrels, which they have replaced throughout most of England and Wales, and parts of Scotland and Ireland. In Italy, the spread of gray squirrels still is restricted to areas near Turin and Genoa, but they already cause considerable damage to local poplar (*Populus* spp.) plantations, and native red squirrels also are being replaced (Wauters et al. 1997). In Canada, gray squirrels were introduced into Stanley Park in Vancouver and have spread over many parts of the lower mainland of British Columbia. There are now populations in Quesnel, Nelson, Bowen Island, and Squamish, B.C. (E. Gonzales, University of Guelph, 1988, pers. comm.). They have also crossed over the border into the United States (D. Fraser, British Columbia Ministry of Environment, Lands and Parks, 1998, pers. comm.). In Calgary, Alberta, gray squirrels escaped from the municipal zoo and have spread throughout the river valley and parks. On Vancouver Island, gray squirrels also escaped from a captive population (Guiguet 1975) and now pose a possible threat to native red squirrels and the endangered Garry oak (*Quercus garryana*) forests.

This paper will outline the history of gray squirrel introductions to Britain and Italy, and describe in detail the effects on native species and forestry operations. Second, it will review the management options currently in use and under evaluation. We then will consider the gray squirrel introduction on Vancouver Island and examine the possible lessons that can be learned from the European experience.

BACKGROUND

Brief History of Eastern Gray Squirrel Introductions to Britain and Italy

The first recorded introduction of eastern gray squirrels (thought to be *S. carolinensis leucotus*) to Britain took place in 1876 in Cheshire, and at least 33 different introductions and translocations had been made by 1930 (Middleton 1930). These occurred from Loch Long in Scotland (1892) to Sandling in Kent (1910), and included an introduction of gray squirrels to Castle Forbes in Ireland (1913). It is thought that, at first, gray squirrels did not spread far from the major points of introduction, but between 1930 and 1945 they dramatically extended their range (Gurnell 1987). Since then gray squirrels have successfully spread to colonize much of England, Wales, and the Scottish lowlands (Gurnell and Pepper 1993), replacing the European red squirrel (Gurnell 1994). Red squirrels are now extinct in southern England, except for some offshore islands such as the Isle of Wight. A few isolated populations can still be found in central England and parts of Wales. The remaining strongholds of the red squirrel are in northern England and Scotland. However, their range is declining even in those areas and careful management of selected conservation areas is required to retain them in northern England in the future (Gurnell and Lurz 1997, Lurz et al. 1998).

In Italy, 4 gray squirrels were introduced in 1946 into the park at Stupinigi, southwest of Turin, and 5 squirrels were released near Genoa in 1966. Surveys have indicated that by 1997, gray squirrels had colonized an area of more than 350 km^2 in northern Italy and that they are replacing the native European red squirrel (Wauters et al. 1997), as is the case in Britain (Gurnell and Pepper 1993) and Ireland (Reilly 1997).

Ecological Replacement of European Red Squirrels by Eastern Gray Squirrels

Understanding the reasons for the replacement of European red squirrels by gray squirrels is vital if effective management strategies are to be formed for the conservation of European red squirrels. Unfortunately, the exact mechanisms of the replacements of red by gray squirrels still are not fully understood. Several hypotheses have been formulated, including interference with red squirrel mating behaviour, direct aggression, competition, and disease (Gurnell 1987, Skelcher 1997). However, the observed replacement of red by gray squirrels indicates that competition based on habitat and food exploitation is the most likely explanation. The presence of large-seeded broadleafed trees, particularly oak, has been suggested as a key factor giving gray squirrels an advantage (Kenward and Holm 1993, Gurnell 1996*b*). Food competition may affect breeding success of red squirrels (Skelcher 1997), and there are indications that it also significantly reduces juvenile red squirrel recruitment (L. Wauters, University of Varese, pers. comm.). The main reasons for this appear to result from adaptations to the different types of habitats within which the 2 species evolved.

European red squirrels are conifer-specialists and probably evolved in boreal forests (Gurnell 1987, Gurnell and Anderson 1996). They are smaller, lighter, and more agile than gray squirrels. This enables them to move easily on the thin branches of coniferous trees and reach food on slender branches (Kenward and Tonkin 1986). In the absence of other squirrel species in continental Europe, red squirrels have colonized all available woodland habitats, including

deciduous woodlands, parks, and gardens. Gray squirrels evolved in eastern North America and are principally adapted to hardwood forests (Gurnell 1987). They are heavier and more terrestrial than red squirrels (Gurnell 1987, Kenward and Holm 1993). There also is evidence that gray squirrels are able to digest acorns more efficiently than European red squirrels (Kenward and Holm 1993). In contrast to European red squirrels, the unrestricted exploitation of this abundant food resource enables them to reach higher densities in hardwood habitats. It has been suggested that gray squirrels colonizing an area will establish themselves in these favourable habitats first and these nuclei will eventually link up to establish gray squirrels over a wider area (Gurnell 1996*b*, Skelcher 1997). Access to oak also has been proposed as a key factor in gray squirrels replacing red squirrels in predominantly conifer forests (Kenward et al. 1998, although see Gurnell 1996*b*). It is believed that red squirrels might find a refuge from gray squirrels only in large conifer forests such as Thetford Forest and Kielder Forest in England (Gurnell and Pepper 1993, Lurz et al. 1995).

Disease

Disease associated with a parapoxvirus is a significant cause of mortality in European red squirrels (Scott et al. 1981, Sainsbury and Ward 1996, Sainsbury et al. 1997). The origin of the virus is not known, but it has been put forward that gray squirrels might be the source, because the disease was not known prior to their introduction about 100 years ago (Sainsbury and Gurnell 1995). However, there has only been 1 case of disease associated with a parapox-like virus described in a gray squirrel in Britain (Duff et al. 1996). Interestingly, viruses found in the western gray squirrel in North America contain structures reminiscent of parapoxviruses (Regenery 1975). In 1995, Sainsbury and Gurnell called for a detailed study of the epidemiology of parapoxvirus transmission and infection. Blood samples subsequently taken from populations throughout Britain have revealed that 60% of gray squirrels tested had been exposed to parapoxvirus infection. This indicates a high percentage of endemic infection in gray squirrels of low pathogenicity and suggests that gray squirrels may be regarded as a reservoir host (see Crouch et al. 1995). However, populations of gray squirrels in northwest England, Scotland, Ireland, and Italy showed no antibodies which could represent differences in the distribution of the virus among populations (Sainsbury et al. submitted; L. Wauters, University of Varese, 1998, pers. comm.).

Economic Damage to Forests through Bark-stripping Behaviour

Gray squirrels are not only a threat to the survival of the European red squirrel in Britain and Italy, they are also a serious problem for commercial hardwood timber production. The location of bark damage on a tree varies among tree species and is probably related to the ease of bark removal and bark thickness, and hence to the growth characteristics of different species. Basal damage (within 1 m of the ground) is the most common type of damage in beech (*Fagus sylvatica*). Crown damage frequently occurs in the main canopy of oaks and many conifers, while stem damage usually occurs between the base and canopy in, for example, sycamore (*Acer pseudoplatanus*), beech, birch (*Betula* spp.), larch (*Larix* spp.), and lodgepole pine (*Pinus contorta*). Damage results in wounds that severely degrade timber quality. Crown damage affects the growth and appearance of the tree; severe crown damage kills it. Damage to the base and stem is cumulative occurring over a number of years. Wounds tend to callous over, hiding the damage until the tree is felled. Trees girdled by excessive stem or basal damage will die. Secondary attacks by rotting or staining fungi can occur at the site of wounds (Dagnall et al. 1998). The damage can be serious enough to kill most of the young trees at a particular site (Shorten 1957, Kenward et al. 1992). Damage occurs principally between May and July when the sap is rising. Trees at 10–40 years of age are most vulnerable (Gurnell and Pepper 1988, Kenward 1989), and bark stripping is most likely to occur when juvenile squirrel density is high (Kenward 1983, Kenward and Parish 1986). It may be triggered by agonistic encounters that cause redirected gnawing behaviour (Gurnell in press). In northern Italy, bark stripping of poplars cultivated for pulp and timber is a particular problem. Wounds can girdle the stem so that the top dies and is blown down by the wind (Currado et al. 1987). The spread of the gray squirrel in Italy and across the Alps into central Europe may have serious implications for forest management, as well as the future survival of the red squirrel in Europe, hence there is an urgent need to control its continued range expansion (Gurnell and Lurz 1997).

REVIEW OF MANAGEMENT OPTIONS

Monitoring

A prerequisite to effective management is the determination of the status and trends of squirrel populations. Monitoring is a key task to achieve this objective. Various methods used in Britain include the collection of presence/absence data by methods such as drey (nest) and cone counts, transect walks, public participation surveys, and the use of hair-tubes (Gurnell and Pepper 1994). The latter are plastic tubes containing bait and blocks with sticky tapes on which visiting squirrels will leave hair. These can be identified using a reference collection of red and gray squirrel hairs and by staining a sample with ink (Teerink 1991, Gurnell and Pepper 1994, Dagnall et al. 1995). Hair-tubes also have been used to determine the current spread of gray squirrels in northern Italy (S. Bertolini, University of Turin, 1998, pers. comm.),

and an attempt has been made to use them in estimating red squirrel population densities in commercial conifer plantations (Garson and Lurz 1998). Population estimates derived from live-trapping (capture–mark–recapture) are labour intensive and thus costl , and additionally require licensing in the case of the European red squirrel. Live-trapping is of limited use as a monitoring method, and is mainly used in intensive research projects.

Management for European Red Squirrel Conservation

Conservation management in Britain involves a number of short- and long-term tactics, which have been reviewed by Gurnell and Pepper (1993) and Lurz and Garson (1997). In the short term these include: the removal of gray squirrels (e.g., by trapping) from selected areas to reduce the effects of competition; if appropriate, the reintroduction or translocation of individual red squirrels into a conservation area to establish or boost a red squirrel population (Venning et al. 1997); and selective supplementary feeding of red squirrels. By providing food only to red squirrels in a mixed red and gray squirrel area it may be possible to offset the competitive advantage gray squirrels have in deciduous habitats. The Forest Commission in the United Kingdom has designed a red-only supplementary food hopper that selects for red squirrels on a weight basis (Pepper 1993). Red squirrels can access the food, but the heavier gray squirrel falls through a see-saw floor before reaching the food container. However, the effectiveness of supplementary food still has to be proven. The efficiency of the supplementary food hopper in keeping out gray squirrels is uncertain, and hoppers could provide foci for the spread of disease. Moreover, the benefits of extra food at the population level are unclear. Research has shown that supplementary food can increase the abundance of North American red squirrels (Sullivan 1990, Klenner and Krebs 1991) and of Douglas squirrels (*Tamiasciurus douglasii*; Sullivan and Sullivan 1982). However, in the case of the European red squirrel, both Holm (1987) and Lurz (1995) reported no benefit from supplementary feeding and concluded that supplementary feeding on its own was not a sufficient management option. Similarly, Shuttleworth (1995) concluded that the removal of gray squirrels may be a better option than supplementary feeding of red squirrels alone, as gray squirrel control reduces the overall "squirrel-pressure" on natural foods.

Long-term management options include designing habitats that benefit red squirrels but deter gray squirrels (Gurnell and Pepper 1993, Gurnell 1994, Lurz et al. 1995). This should include felling and restocking programs that maintain forest cover, connectivity, and a continuous seed-food supply. Modelling approaches can assist here to predict the spatial distribution of both squirrel species in relation to gray squirrel spread and habitat composition (Armitage et al. 1997, Rushton et al. 1997).

Damage Control

Damage control in Britain has been reviewed by Gurnell and Pepper (1998), Dagnall et al. (1998) and, more recently, Gurnell (in press). It is not possible to eradicate gray squirrels from Britain, but it is still possible to do so in Italy before their distribution becomes too large. Gray squirrel numbers can be reduced locally and in the short term by targeting control. Since 1973, the most cost-effective method in Britain has been the use of warfarin poison from special hoppers that keep out non-target species (Pepper 1990). This is legally controlled by the Warfarin Order 1973, the Wildlife and Countryside Act 1981, and the Control of Pesticides Regulations 1986. There is no significant secondary poisoning hazard to birds such as the tawny owl (*Strix aluco*) from warfarin in prey animals, but weasels (*Mustela nivalis*) could be at risk if they feed exclusively on contaminated animals (Townsend et al. 1984). It is illegal to use warfarin for gray squirrel control where red squirrels are at risk. The use of poison is not allowed in Italy. Live-trapping is also effective but more labour intensive and costly (Gurnell in press); it may be the only method available if warfarin is banned from use under European law (Annex A of the European Union Plant Protection Directive; H. Pepper, Forestry Commission, pers. comm.).

A more recent approach involves synthetic oral immunocontraception. In this technique, DNA coding for specific sperm or egg antigens involved in fertilization are introduced into the animal via oral vaccination. This causes the body to produce antibodies against the targeted egg or sperm proteins, thereby creating an immunocontraceptive barrier and rendering the host sterile (Moore 1997). The success of immunosterilization techniques in the case of the gray squirrel depends on the identification of suitable reproductive antigens that are species-specific, an effective delivery system, and an understanding of the ecology of the animal (H. Moore, University of Sheffield, 1998, pers. comm.). It could be several years before this form of control becomes available.

EASTERN GRAY SQUIRRELS IN BRITISH COLUMBIA

History of Eastern Gray Squirrel

Introduction to Vancouver Island

The eastern gray squirrel introduction on Vancouver Island originated at Metchosin in 1966 (Guiguet 1975), when 2 females and 1 male imported from Ontario escaped from a game farm. At first the population remained fairly concentrated near the site of the release, but over the past 15 years the species has increased its range dramatically. Gray squirrels have now spread throughout the southern portion of Vancouver Island, including northward through Goldstream Provincial Park and beyond. In recent years, the range of the animals also has expanded through Esquimalt to beyond

Sooke, and along the Saanich Peninsula almost to the ferry terminal at Swartz Bay. As the population density around Victoria has increased, more and more pest controllers have been called upon to trap the animals. In some cases the captured animals are re-released farther north on the island, thus exacerbating the situation. Gray squirrels can now also be found on the north end of the Malahat at Bamberton, Duncan, and Nanaimo (D. Fraser, British Columbia Ministry of Environment, Lands and Parks, 1998, pers. comm.). The continued release of gray squirrels and the favourable habitat over large parts of Vancouver Island make a continued and rapid expansion of the gray squirrel population appear unavoidable.

Displacement of North American Red Squirrels

The North American red squirrel evolved in the conifer forests of North America (Obbard 1987, Gurnell and Anderson 1996) and, like the European red squirrel, it is a conifer-specialist. North American and European red squirrels fill a similar ecological niche on their respective continents, and, therefore, one might expect that they will be impacted by the invasion of gray squirrels in a similar fashion; that is, displacement would occur in hardwood and mixed woodlands and in urban environments.

Since North American red squirrels are largely territorial and dependent on defending middens, they can be more aggressive than eastern gray squirrels and there have been reports of red squirrels chasing off gray squirrels. Although this may be true on an individual basis, gray squirrels can reach higher densities in mixed and deciduous habitats, and red squirrels may become outnumbered and eventually replaced. Additionally, in deciduous forests (the habitat type in which they are most likely to suffer from the presence of gray squirrels), red squirrels cease to defend territories and adopt a social organization based on overlapping home ranges (Gurnell 1987). Both the gray squirrel and the European red squirrel live in overlapping home ranges and, as we have seen in the United Kingdom and Italy, this system clearly allows displacement of red squirrels by gray squirrels (Gurnell 1987). In fact, a displacement of red by gray squirrels on Vancouver Island has already happened in habitats other than conifer woods in and around Victoria (D. Fraser, British Columbia Ministry of Environment, Lands and Parks, 1998, pers. comm.).

The British situation has shown that disease may play a role in the displacement of red squirrels by gray squirrels. Not only may gray squirrels aid the spread of a disease already existing in a native population, they also may introduce a new disease to which native populations have not evolved immunity.

Impacts on the Garry Oak Ecosystem

As discussed above, gray squirrels are hardwood-specialists and are likely to do best in hardwood and mixed woodland types on Vancouver Island. The endangered Garry oak ecosystems are, therefore, prime candidates for habitats where gray squirrels may replace North American red squirrels and reach higher densities than those of the red squirrel populations they have replaced.

In British Columbia, the Garry oak ecosystem is restricted mainly to the southeast coast of Vancouver Island and the southern Gulf Islands. Urban development and the encroachment of another alien species, Scotch broom (*Cytisus scoparius*), have been important causes of ecosystem loss in the past (Erickson 1993). The Garry oak ecosystem has been placed on the provincial Red List in view of its threatened status within the province, and it has become one of the rarest habitat types in Canada. The Garry oak is the only oak native to British Columbia. On Vancouver Island, Garry oak ecosystems have existed without the presence of the hardwood-specialist gray squirrel. There are several possible repercussions of the establishment of gray squirrel populations in Garry oak stands:

Prevention of natural regeneration

High densities of gray squirrels in hardwood forests could prevent natural regeneration (e.g., Shaw 1968, Gill et al. 1995). Even though gray squirrels scatterhoard seeds during autumn and are sometimes cited as agents for seed dispersal (e.g., Mellanby 1968), they cut out the radicle of acorns from the white oak group (which includes Garry oak) soon after they fall to the ground, thus preventing their germination (Fox 1982, Pigott et al. 1991). "Notched" acorns have been observed in Garry oak ecosystems on Vancouver Island (M. Fuchs, Foxtree Ecological Consulting, 1997, pers. comm.). Further evidence comes from England, where gray squirrels have been known to completely destroy germinating acorns (Pigott et al. 1991). In many urban parks and gardens in the United Kingdom (where there is no squirrel control) young trees seldom reach 30 years old (Gill et al. 1995). Thus, it seems likely that gray squirrels will affect recruitment of younger oak trees. In fact, diminished oak regeneration already has been observed in some Vancouver Island stands (K. Stewart, Garry Oak Meadow Preservation Society, 1997, pers. comm.). It is not clear at this time whether gray squirrels are the cause of this trend or a contributing factor.

Bark-stripping damage to trees

Research in Britain has shown that young oak trees (10–40 years old) are among the species most frequently bark-stripped. With all the other pressure on the endangered Garry oak ecosystem, this additional threat could prove to have severe consequences on the perpetuation of this habitat. Damage to oaks often occurs in the crown, and detailed surveys are required to see whether damage is occurring where gray squirrels have colonized Garry oak forests.

Predation

There is an ever-growing awareness of the fact that squirrels often prey on other animal species as well as plants (e.g., O'Donaghue 1994). Nesting birds and their offspring are particularly vulnerable (Sullivan 1991). Although the native red squirrel will exhibit predatory behaviour, higher gray squirrel densities in Garry oak ecosystems likely will result in higher predation rates on other species using the same habitat. As the Garry oak community on Vancouver Island has not developed in the presence of the gray squirrel, the ecological impact may be severe.

Competition with native species

Besides competing with and replacing the native red squirrel, competition from gray squirrels may affect the numbers and distribution of other acorn-eating species.

Potential Consequences of the Gray Squirrel Introduction on Vancouver Island

To summarize, we predict that the impacts and the future of gray squirrels on Vancouver Island are as follows:

- Eastern gray squirrel populations will expand further, with the speed of increase varying from year to year, depending on factors like weather, food supply, disease, and translocation by humans.
- The initial expansion will be mainly into deciduous and mixed woodland and urban environments.
- Gray squirrels will replace North American red squirrels in the above habitats. The southeastern part of Vancouver Island contains a reasonable proportion of forests with non-conifer species and urban areas in which this displacement is likely to occur. This predicted shift from red squirrels to gray squirrels already has been observed in some areas of Vancouver Island.
- Gray squirrel populations will reach higher population densities in these habitats than North American red squirrels populations.
- The Garry oak ecosystem will suffer especially from the invasion of gray squirrels as it is already under threat from other pressures and gray squirrels can reach particularly high densities in this type of habitat. Gray squirrels may affect the ecosystem through prevention of regeneration, bark stripping damage, predation, replacing native red squirrels, and food competition with other native seed-eaters.
- Potentially higher densities of gray squirrels compared to North American red squirrels may result in an increase of squirrels raiding bird feeders, bird nests, and fruit crops in urban areas and gardens.

FUTURE RESEARCH AND MANAGEMENT

We wish to add a cautionary note here, as the gray squirrel introductions to Europe and Vancouver Island cannot be directly compared because of differences in habitats and climate. Further, the European and North American red squirrels are different genera. However, we believe that there are enough similarities to be concerned about the impacts on Vancouver Island's native flora and fauna. This warrants further research into the Vancouver Island introduction.

What needs to be done? There are several lines of research that should be carried out. For example, the changing distribution of gray and red squirrels on Vancouver Island needs to be mapped, and the future distribution of gray squirrels predicted using modelling methods. Exclosure studies can be used to study the effects of gray squirrels on native fauna and flora, including natural regeneration in Garry oak stands. Tree damage surveys also need to be carried out and experimental field studies on red-gray interactions conducted, together with studies on the possible impact of disease. Research should be undertaken in a timely fashion while the gray squirrel expansion is not complete and "controls" still are available. Perhaps equally important is to raise public awareness of the problem through various educational programs. This is particularly so if local authorities recommend in the future some level of gray squirrel control or other intervention management techniques (see Gurnell 1996*a*).

Probably the most important lesson to learn from the U.K. experience is to investigate the potential problems as soon as possible after the introductions, and to explore potential management options. It is fortunate that gray squirrels have been present on Vancouver Island for only 30 years, and much of the island has yet to be colonized. However, some impacts have been already observed and there may be others that have so far gone unnoticed. The potential threat from gray squirrels on Vancouver Island should receive due attention as soon as possible.

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State of Regional Parks: An Ecological Perspective

Tracy L. Fleming
Environmental Protection and Conservation Specialist
Capital Regional District Parks
490 Atkins Road, Victoria, BC, V9B 2Z8, Canada
tfleming@crd.bc.ca

ABSTRACT

Population and development pressures in the Capital Regional District (southern Vancouver Island, British Columbia, Canada) are threatening sensitive ecosystems. A strategy of protecting and maintaining existing regional parks, and acquiring land is recommended to protect these ecosystems.

Key words: Capital Regional District, Garry oak, parks, protection, urban development, Victoria.

The Capital Regional District (CRD) is located on southern Vancouver Island, British Columbia, and includes the city of Victoria. The CRD is experiencing rapid population growth along with its associated land development. The rate of population growth is expected to more than double by the year 2020. Some of the main attractions for people to our region are that it is a nice place to live, with a moderate, "Mediterranean" climate (part of that rain forest mystique), and that we are surrounded by the stunning Olympic Mountain range.

The mild climate, milder than anywhere else in Canada, has also created an environment suitable to ecosystems that have most of their range further south in the U.S. Pacific Northwest. This northern extent of their range, plus encroaching development pressures, makes many species and ecosystems rare (even globally rare) and critically imperilled in British Columbia and Canada. The Garry oak ecosystems of this region are prime examples of habitats at the northern extent of their range that have been largely lost to development pressures.

Indeed many of these ecosystems in which species of conservation concern occur are fragments or remnants of their previous abundance, and now are either protected in parks or are on private lands that are slated for development. This case is especially common in the southeastern and southern part of the region. Ecosystems converted to residential developments and urban sprawl are irreversibly changed. In contrast, ecosystems in the western part of the region are largely in managed tree farms, where the original large tracts of old-growth forests remain only as fragments. Areas that have been logged have the potential to re-establish forests; however, active forest management precludes old growth for a long time.

CRD Parks is the CRD government's parks department. The parks managed by CRD Parks make up about one-half of the total area with protected status in the region (provincial parks make up about one-quarter) or about 8,200 ha. As with most park agencies, the biggest challenge is to ensure that the ecosystem functions, remnant natural and semi-natural areas, marine shorelines, and the plant and animal populations they support remain intact and viable over the long term. Thus the role of CRD Parks in protecting remnant areas containing threatened organisms and ecosystems has become critical in our region.

In July 1998, CRD Parks produced a report called *State of Regional Parks: An Ecological Perspective* to research and report on: (1) the current health of ecosystems within regional parks; (2) the key threats to those ecosystems, and (3) the remnant natural areas that are currently outside parks protection (Fleming 1998).

This report was produced as a background document for the *Draft Master Plan* (Capital Regional District Parks 1998) which is currently available for public and internal review. Two strategic direction statements in the master plan recognize the importance of natural areas protection: (1) protect and maintain the natural environment in existing regional parks; and (2) acquire additional lands for regional parks.

Some of the most threatened ecosystems or habitats in the region can still be found within CRD Parks, such as Garry oak meadows, coastal bluffs, wetlands, and riparian and old-growth forests. The protection of the natural areas within parks is challenged by the development of facilities, trails, and parking lots to serve the increasing numbers of visitors. Wear and tear on trails, dogs running unchecked, and garbage are all potential conflicts with natural area protection. Limiting some types of recreation in sensitive areas, public education programs, environmental impact

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

assessments, biological inventories, and thoughtful planning are all proactive ways of minimizing damages and threats to ecosystems in regional parks.

One of the key threats to parks is not knowing what species and ecosystems of conservation concern are in the park. A classic scenario, for which all park agencies have examples, is a trail being constructed next to or even over a rare plant or population of rare plants. Environmental impact assessments for all parks development projects are needed to avoid such scenarios. Also, inventory information needs to be communicated to the operational staff of parks to increase their level of knowledge about the rare and threatened plants that live precarious existences near trails and facilities.

In the summer of 1998, CRD Parks staff discovered a previously unknown location of a population of deltoid balsamroot (*Balsamorhiza deltoidea*). This endangered plant was found because an ecological inventory was being conducted in the park. This population is currently threatened by a high cover of the invasive plant Scotch broom (*Cytisus scoparius*), which is competing for light, moisture, and nutrients. Because we now know that this sensitive plant grows in our park, we will make every effort to protect it.

All of our regional parks have been invaded by nonnative or exotic plants, largely garden escapes. The mild climate of our region has allowed some shrubs (e.g., Scotch broom, daphne [*Daphne laureola*], holly [*Ilex aquifolium*]) to "naturalize" in indigenous ecosystems. Scotch broom has become successful in open areas like Garry oak meadows causing ecosystem changes by increasing fixed nitrogen in the soil which encourages a preponderance of agronomic grasses, as well as unfavourable conditions for the native wildflowers. Recent CRD Parks guidelines for the management of invasive plant species have recognized the vast challenge of eradicating plants like Scotch broom. Therefore, our strategic approach to their removal and control includes 2 broad guidelines: (1) prioritize the exotic species targeted for removal; and (2) prioritize those parks and ecosystems that have the highest conservation concern, and remove priority exotic species.

It is becoming increasingly popular for residential developments to be built beside parks. In fact, many developers and local governments have advertised proximity to natural areas as a selling feature for housing. All sorts of threats to the species and ecosystems within parks result. Noticeably, parts of ecosystems are lost because ecosystem boundaries do not stop at park boundaries. Park boundaries are not based in nature—they are artificial. The ecosystems and watersheds are rarely protected in whole. The creeks and rivers are only partially protected by parks, and the waters that flow in them are affected by what is happening in developed areas upstream. Other key threats to the natural areas within parks that result from adjacent land development are: (1) increased numbers of personal access trails from private properties which result in fragmented ecosystems; (2) expansion of backyards into the park and dumping of garden waste, which provides new invasion points for nonnative plants (and animals); and (3) removal of natural resources, plants, and animals from the park, such as gathering mosses for hanging baskets, picking salal for flower arrangements, and picking mushrooms and wildflowers.

CRD Parks works closely with municipal governments and developers to reach compromises that add parkland as buffers around regional park boundaries. This kind of arrangement can help to mitigate the conflicts between ecological protection and development. To protect CRD Parks interests on land beside an existing regional park, several options exist such as easements, special permits, private land stewardship, conservation covenants, and community group efforts (Capital Regional District Parks 1998).

Opportunities still exist to acquire important remnant ecosystems in the CRD. The *State of Regional Parks* report (Fleming 1998) provides direction for park land acquisitions from an ecological perspective. First and foremost, ecosystems that extend beyond present park boundaries have been identified as targets for acquisition. Connecting natural areas is also an acquisition priority where private lands can be acquired, or protected by some other means, to strategically link one regional park (or protected area of another jurisdiction) to another park. Riparian corridors are prime examples of critical habitat in need of protection that can connect protected areas in parks.

Few opportunities remain to acquire remnants of Garry oak ecosystems. Much of what is now urban Victoria once was covered by open Garry oak savannahs and meadows. These deeper soil Garry oak ecosystems have been converted to manicured lawns surrounding the oak trees. The Garry oaks and wildflower meadows of thin soil rock outcroppings are also threatened as residential development pushes up onto the hillsides. Remnants will be targeted for protection, with the knowledge that it is increasingly urgent to protect the Garry oak ecosystems already within regional parks.

The most important recent acquisition by CRD Parks is the Sooke Hills Wilderness Regional Park Reserve; acquiring this tract of land has doubled our regional park land base. Sooke Hills is just northwest of the western extent of development associated with the Greater Victoria area. It had been held within Victoria's water supply lands, and as a regional park continues to be a buffer for the drinking water supply of the city. Although much of the Sooke Hills is second-growth forests, it also contains large tracts of older mature and old-growth forests. This wilderness area is important wildlife habitat where black bears, cougars, and wolves live. No other protected areas of this size, about 4,000 ha, occur on southern Vancouver Island, and some wildlife species require these large protected landscapes to

maintain viable populations.

Recent research in the Sooke Hills illustrates the need for large contiguous tracts of natural lands. Last summer a team of researchers from the University of Victoria found marbled murrelets flying around in old-growth forests in the Sooke Hills (Burger et al. 1999). The British Columbia Conservation Data Centre (CDC) has Red-listed these little birds, meaning that their breeding population has been extirpated, endangered, or threatened in this province, and they likely occur only as a diffuse, moving, nonbreeding population (B.C. Conservation Data Centre 1999). These shy sea birds are unique in that they nest on the large mossy branches of very old trees and fly great distances between forest and ocean habitats. With further research, the University of Victoria team hopes to find some nests in these "hot spots" where marbled murrelets are active in the Sooke Hills.

The gaps in protection of remnant natural areas were analyzed as part of the *State of Regional Parks* report (Fleming 1998). Using the Sensitive Ecosystems Inventory (SEI) compiled and mapped by the CDC (B.C. Conservation Data Centre 1996), those ecosystems sensitive to human disturbances, such as those containing rare and threatened species of flora and fauna that are not in protection status, were flagged, described, and indicated on maps (Fleming 1998). Garry oak ecosystems, old-growth forest, wetlands, and riparian areas were identified as high priorities for protection in the region. Many of these unprotected areas also appear as areas of interest to CRD Parks (Capital Regional District Parks 1998).

CRD Parks feels that we still have the opportunity to influence the development pattern in the western part of the region by protecting the remaining best examples of ecosystems and landscapes.

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Status of Fish Habitat in East Coast Vancouver Island Watersheds

George E. Reid
Fisheries Section, British Columbia Ministry of Environment, Lands and Parks
2080 Labieux Road, Nanaimo, BC, V9T 6J9, Canada
George.Reid@gems8.gov.bc.ca

Tracy A. Michalski
Fisheries Section, British Columbia Ministry of Environment, Lands and Parks
2080 Labieux Road, Nanaimo, BC, V9T 6J9, Canada

Thomas Reid
c/o George Reid, Fisheries Section, British Columbia Ministry of Environment, Lands and Parks
2080 Labieux Road, Nanaimo, BC, V9T 6J9, Canada

ABSTRACT

The Vancouver Island boundary of the Georgia Basin is 1 of the fastest growing areas in British Columbia. The area has an abundance of small, second-order streams, most of which originate in private forested lands, then run through rural and urban environments. We examined the instream habitat of 14 watersheds in that area. We found reduced pool area in 41% of the watersheds, a lack of large woody debris (LWD) in 93%, reduced instream cover in 50%, and excessive fine sediments in 88%. Our analysis shows summer low-flow problems in 155 of 165 streams examined. We found an average of 25.4% development in the 14 watersheds as a result of urban, agricultural, golf course, and/or rural residential land use. We make a number of suggestions for improving the protection of east coast streams, including more public involvement in watershed planning and the development of Official Community Plans (OCPs) and bylaws that provide for the protection of small stream habitat during and after development. We also recommend how volunteers may become involved in long-term protection and restoration of their watersheds.

Key words: fish, habitat restoration, land use, public involvement, small streams, Vancouver Island.

The Georgia Basin of British Columbia is 1 of the fastest growing areas on the continent. More than two-thirds of the population of the province live there, yet the region makes up less than 3% of British Columbia's total area (Owen 1994). By the year 2020, Vancouver Island's population is expected to double, expanding the population to more than 850,000 along the narrow east coast of the island (Owen 1994, B.C. MFCR 1997). The impact on small stream fish habitat, given the island's sheer physical constraints, will be significant.

Over the past decade, the involvement of volunteers in a variety of stewardship initiatives has increased significantly due to growing public concern over deteriorating natural habitats. One government program providing funding and technical support to stewardship projects in the Georgia Basin is the Urban Salmon Habitat Program (USHP). Since its inception in 1995, the program has funded 77 different projects on Vancouver Island, focusing on increasing community awareness, habitat and fish assessment, and stream and riparian restoration.

We insist that stewardship groups undertake fish habitat assessments to identify and prioritize the factors limiting fish production before any restoration is contemplated. Some of the elements contributing to salmonid habitat include: large logs that provide instream rearing during the summer and winter; suitable-size, clean gravel for spawning; and boulders, logs, and roots that provide escape cover (Lister and Genoe 1970, Bustard and Narver 1975, Griffith 1980, Wesche and Rechard 1980, Bisson et al. 1981, Bjornn and Reiser 1991). We have volunteers assess their streams during the low-flow period (August–September). Summer flows below 10% of the mean annual discharge (MAD) result in reduced stream carrying capacity (Tennant 1976). Wickett (1951) showed that smolt production is positively correlated with stream discharge.

We also have volunteers determine the land use adjacent to the riparian zones. Poor land-use practices can result in vegetative clearing; soil compaction; stream channelization; riparian corridor alterations; chemical pollution; and changes in flow regime (Benke et al. 1981, Booth and Reinelt 1993, City of Olympia 1994). Urban development is of special concern due to the impact of impervious surfaces on small stream habitat. A number of studies show that stream degradation occurs when the amount of impervious area in a watershed exceeds 10% (Lucchetti and Fuerstenberg 1993, Precision Identification Biological Consultants 1997, Green Mountain Institute 1998).

In this study we examined the current state of fish habitat

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Table 1. Streams assessed by stewardship groups in 1995 and 1996 using the Urban Salmon Habitat Program Assessment Procedures.

Creek	Stewardship group	Area	Year stream assessed
Ayum Creek	Society for the Protection of Ayum Creek	Sooke	1996
Beach Creek	Qualicum Beach Streamkeepers	Qualicum Beach	1996
Bear Creek	Oyster River Watershed Management Committee	Campbell River	1996
Fairways Creek	Oyster River Watershed Management Committee	Campbell River	1996
Kingfisher Creek	Haig-Brown Kingfisher Creek Society	Campbell River	1996
Little River	Little River Enhancement Society	Comox	1996
Little Oyster River	Oyster River Watershed Management Committee	Campbell River	1996
Nile Creek	Nile Creek Enhancement Society	Bowser	1996
Piercy Creek	Millard/Piercy Watershed Stewards	Courtenay	1996
Scales Creek	Little River Enhancement Society	Comox	1996
Simms Creek	Campbell River Fish and Wildlife Association; Campbell River Guides Association	Campbell River	1996
Thatcher Creek	Nanaimo Fish and Game Protective Association	Nanaimo	1995
Woodhus Creek	Oyster River Watershed Management Committee	Campbell River	1996
Woods Creek	Storie Creek Golf and Recreation Society	Campbell River	1996

and land use in 14 study streams, and flows in 165 watersheds on the east coast of Vancouver Island. We found severe impacts to fish habitat, due to a variety of factors including poor land-use practices. We discuss the implications for salmonid habitat management in small streams including implementing comprehensive new strategies such as publicly supported land-use plans to protect small stream fish habitat. We also recommend stewardship groups and local governments share the responsibility of long-term development, monitoring, and protection of their streams and that senior governments enforce the legislation governing these activities. Finally, we recommend community-focused stream restoration and public awareness programs.

METHODS

Stewardship groups completed 14 USHP fish habitat assessments between 1995 and 1996 using an assessment procedure developed by the Urban Salmon Habitat Program on Vancouver Island (Table 1; Fig. 1; Michalski et al. 1995). The procedure involves collecting data for a number of parameters including: percentage of pools; percentage of instream cover; large woody debris/bankfull channel width (LWD/BCW); percentage of fines; and percentage of wetted area along the entire length of the stream, including tributaries. Instream cover includes large woody debris, boulders, undercut banks, and instream vegetation. We entered the stewardship data into a master database and produced a number of summaries and reach roll-ups for the streams and parameters sampled. We then produced island roll-ups to compare the fish habitat parameters to biostandards for healthy streams (Table 2; Johnston and Slaney 1996).

Tennant (1976) noted that stream width, mean discharge, depth, and velocity vary as a function of the mean annual discharge. He defined MAD flows in terms of impacts to fish habitat (Table 3). We used a provincially modified version of the Tennant Method to analyze 165 east coast Vancouver Island streams and then classified these streams according to Tennant's criteria.

We determined the type and percentage of land use in each of the 14 study watersheds by defining watershed boundaries on large-scale aerial photographs and maps and then digitizing each land use type. (Table 4). We used 1:11,000-scale aerial photographs and 1:5,000-scale cadastral maps for areas where aerial photographs were not available.

We calculated the total amount of impervious area after

Table 2. Biostandards for instream fish habitat parameters (modified from Johnston and Slaney 1996).

Habitat Parameter	Biostandard	Rating
Pools	>55% of reach	Good
	40–55% of reach	Fair
	<40% of reach	Poor
Instream cover	>20% cover	Good
	6–20% cover	Fair
	0–5% cover	Poor
LWD/BCW[a]	>2 pieces/BCW	Good
	1–2 pieces/BCW	Fair
	<1 piece/BCW	Poor
Fines	<10% of area sampled	Good
	10–20% of area sampled	Fair
	>20% of area sampled	Poor
Wetted area	>90%	Good
	70–90%	Fair
	<70%	Poor

[a] LWD = Large Woody Debris; BCW = Bankfull Channel Width.

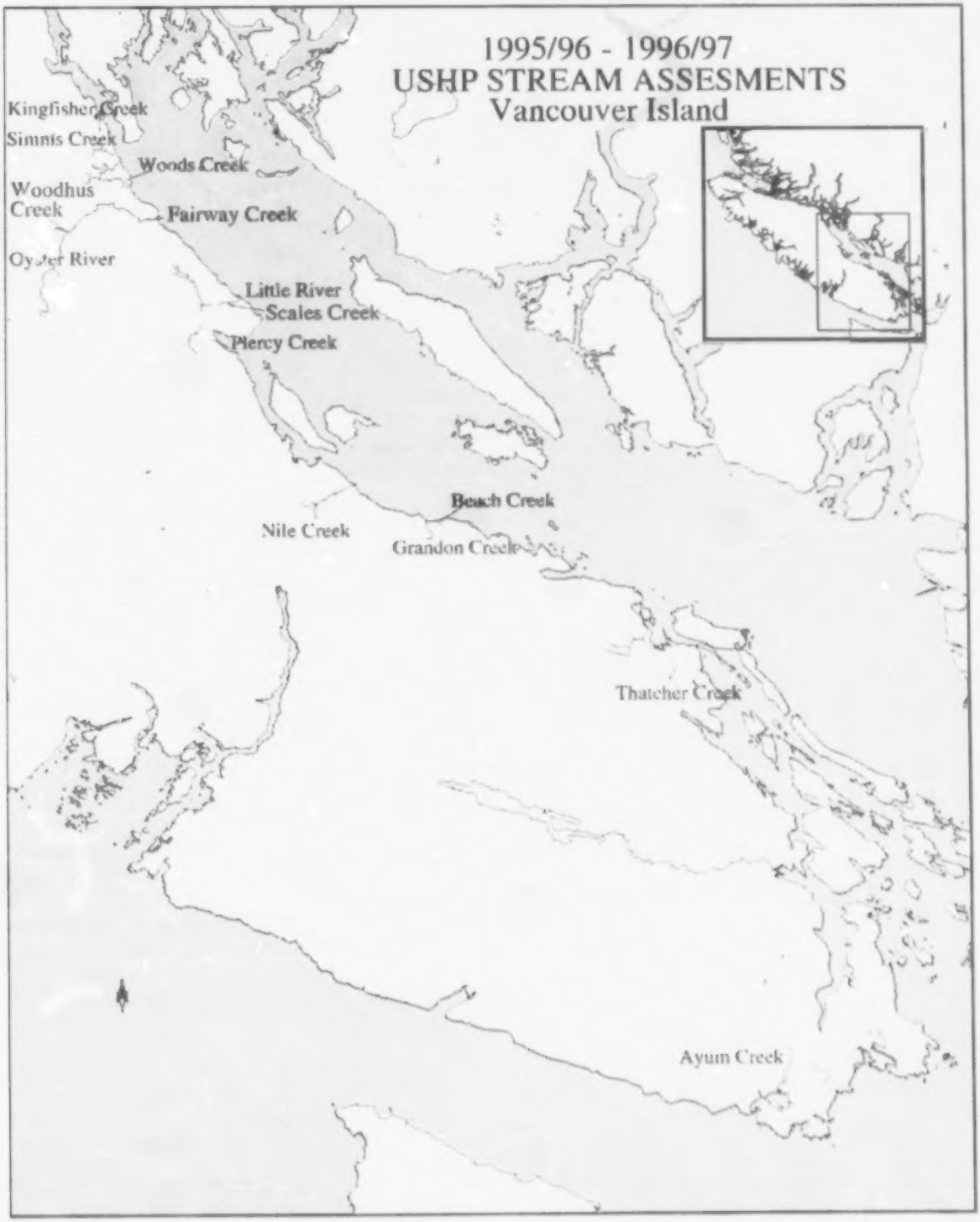


Figure 1. Study location.

Table 3. Instream flow requirements based on the modified Tennant (Montana) Method.

Flows (% Mean Annual Discharge)	Description
30-60% MAD	Excellent spawning/rearing
20-30% MAD	Good spawning/rearing
10-20% MAD	Fair spawning/rearing
5-10% MAD	Poor spawning/rearing
<5% MAD	Severely degraded spawning/rearing

we determined the amount of urban development. Total impervious area is composed of the amount of rooftops plus the amount of transportation links in a watershed (Schueler 1994). Researchers in Washington State showed that road-related impervious area comprised 64–70% of total impervious area in 11 residential, multifamily, and commercial areas (City of Olympia 1994). We used a conservative estimate of 60% to calculate the road component of total impervious area. The impervious area in Washington State associated with rooftops in medium-density, single-family homes ranged from 25% to nearly 60% (Schueler 1994). We used the median of 40% to account for the impervious area related to rooftops. We calculated the total impervious area for the 14 study watersheds as:

Impervious Area (m^2) = $I_{roof} + I_{road}$

% Impervious Area = Impervious Area/Watershed Area

where: I_{road} = 60% of Urban Area; I_{roof} = 40% x (Urban Area - Road Area).

Researchers in Washington developed a stream quality index, which we used to classify the status of stream quality based on the amount of impervious area in each watershed (Table 5; Schueler 1994).

RESULTS

We calculated the average percentage pool area for all assessed streams as approximately 50%, slightly below the biostandard of 55% (Fig. 2). A total of 6 streams, including Ayum, Kingfisher, Piercy, Scales, Simms, and Woodhus creeks, did not meet the biostandard. The remaining 8 streams ranged from 58% pool habitat in Beach Creek to 90% in Woods Creek.

Table 5. Stream quality index based on the amount of impervious area (modified from Schueler 1994).

Percent of impervious cover	Status of stream quality
1–4%	Minimum impact
5–10%	Stressed
11–25%	Impacted
26–100%	Degraded

Table 4. Land use categories and their descriptions determined for the study watersheds on the east coast of Vancouver Island.

Land Use	Description
Forest	Tree-covered land.
Rural residential	Residential and other buildings in low density.
Urban	Cities, towns, and villages, as well as isolated areas such as manufacturing plants, rail yards, industrial areas etc.
Agriculture	Land-based agricultural activities.
Exposed/logged	
Golf course	Fairways and greens.

The average percent instream cover found in 13 of the study streams was approximately 21%, slightly above the biostandard of 20% (Fig. 2). Thirty-eight percent of the streams had >20% instream cover, while 62% of the streams fell below the biostandard. None of the study streams had <3% instream cover, although the Little Oyster River fell on that border.

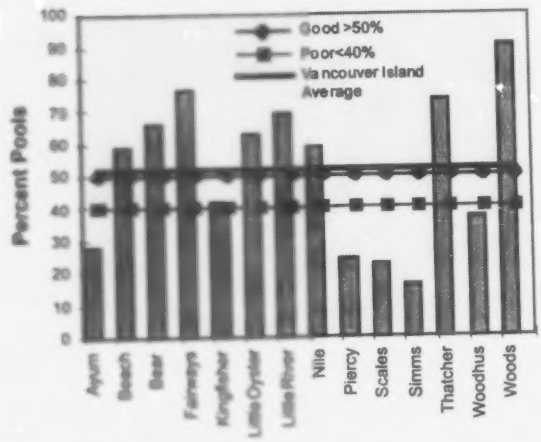
The biostandard for LWD/BCW is 2 (Johnston and Slaney 1996). We calculated the average number of pieces of LWD/BCW at 0.78 (Fig. 2). We found LWD/BCW limiting in all but 1 stream. Ten of the study streams contained <0.5 pieces of LWD/BCW, while 1 creek had <0.1 pieces.

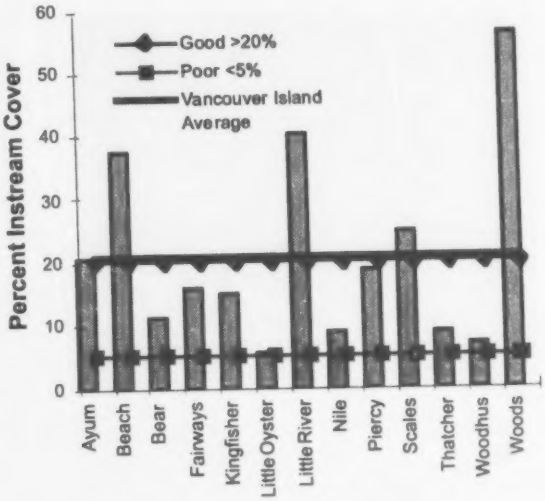
We analyzed the percentage of fines in 8 of the 14 study streams (Fig. 2). The remaining streams had incomplete data for this parameter. We calculated the average percentage fines at 50.2%, well above the biostandard of 10%. Only Ayum Creek had <10% fines, as required for unimpaired salmonid production. We found 95% fines in Beach Creek; >86% fines in Thatcher Creek; and 68% fines in Little River.

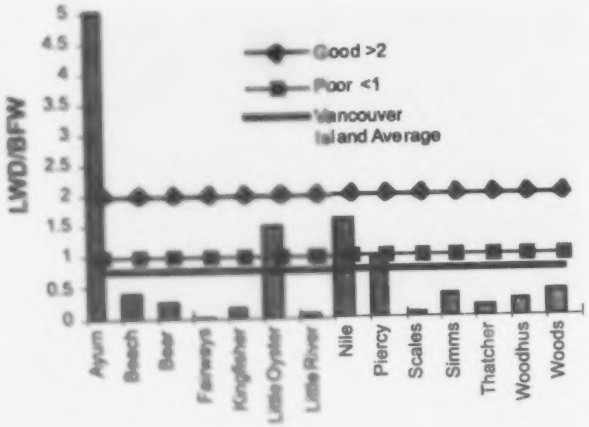
We did not find any streams that met the 90% biostandard for wetted area (Fig. 2). The average percentage wetted area was 64% for the 13 streams examined. Only Fairways, Kingfisher, and Woodhus creeks had >70% of the channel wetted during the assessment. The percentage of wetted area in the remaining 10 creeks varied from 30% in Scales Creek to 69% in Simms Creek.

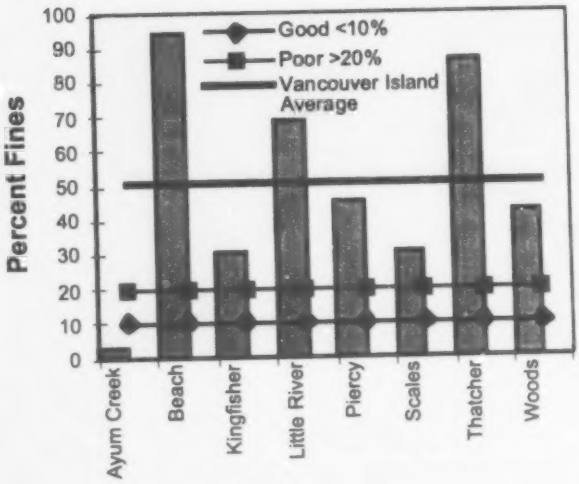
Eighty-two percent of the 165 streams had flows <5% MAD (Fig. 3). A further 11.5% of these streams had flows between 5 and 10% MAD, for a total of 93% with degraded spawning and rearing habitat. Only 6% of the streams we examined had summer flows >10% MAD. Seven of our 14 study streams had a Water Allocation Plan. Five of these streams had flows <5% MAD; 1 had a summer flow of 8% MAD; and 1 had a flow >20% MAD (Table 6).

Forest land was the predominant land-use type in all watersheds (Table 7). The percentage of forested area varied from 42.4% in the Piercy Creek watershed to 99% in the Woodhus Creek watershed. We found 12 streams with logged









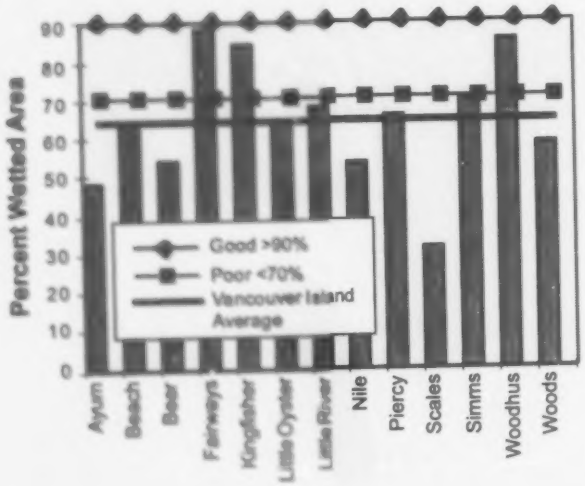


Figure 2. Percent pool area, percent instream cover, pieces of Large Woody Debris/Bankfull Channel Width; percent fines; and percent wetted area in fourteen Vancouver Island streams assessed using the Urban Salmon Habitat Program Assessment Procedures between 1995 and 1996.

Table 6. Mean Annual Discharge (MAD) and low flow measurements for 7 east coast Vancouver Island streams with Water Allocation Plans.

Stream	MAD (L/sec)	Critical flow (L/sec)	% MAD	Classification[a]
Ayum Creek	411	0.9864	0.48	Severely degraded
Bear Creek	310	0	0	Severely degraded
Beach Creek	171	0	0	Severely degraded
Little River	332	26.56	8	Poor
Little Oyster River	1,600	4.96	0.31	Severely degraded
Nile Creek	985	197.99	20.1	Good
Woodhus Creek	1,500	9.9	0.66	Severely degraded

[a] According to modified Tennant Method.

or exposed areas. Seventeen percent of the Woods Creek watershed had logging, which was the highest of all streams examined. The amount of urban development in the study watersheds varied from 0% in the Little Oyster River to 25.8% in Simms Creek. Most watersheds in the study had <10% of the watershed in urban development, but Beach, Piercy, and Simms creeks had >20%. Six of the study streams contained agriculture in their watersheds. The amount of agricultural land use varied from 7.4% in the Beach Creek watershed to 32.7% in the Thatcher Creek watershed. We found golf courses in 6 of the 14 watersheds studied. The percentage of golf course development varied from 3.3% in the Fairways Creek watershed to 7.5% in the Woods Creek watershed. Woods, Bear, and Kingfisher watersheds were the only ones with rural residential areas.

Impervious area varied between 0% in the Little Oyster River watershed to 19.6% in the Simms Creek watershed (Table 8). The average amount of impervious area in all study streams combined was almost 8%. Ayum, Bear, Little Oyster, Nile, and Woodhus watersheds had <4% impervious area. These watersheds have been classified as having minimal impact from impervious area (Table 9). Kingfisher, Little River/Scales, Thatcher, and Woods watersheds had 5–10% impervious area (Fig. 4). We classified the status of stream quality in these watersheds as stressed. Beach, Fairways, Piercy, and Simms watersheds had 11–25% impervious area and we classified these streams as impacted as a result of impervious area.

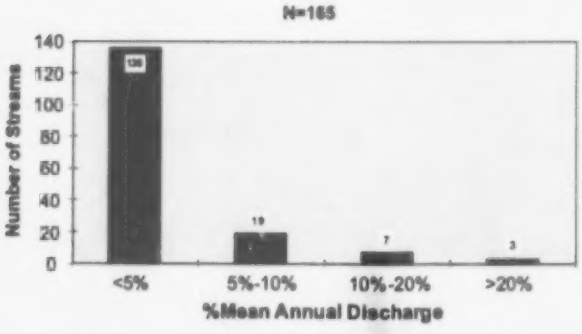


Figure 3. Mean Annual Discharge (MAD) for Vancouver Island streams for which a Water Allocation Plan has been completed.

DISCUSSION

The Problem

Vancouver Island is fortunate in that the vast majority of the region is forested. Owen (1994) calculated 84% of the east coast of Vancouver Island as forest land, 9% urban, and 3% agricultural. Forest land made up an average of 68.2% for all of the watersheds we examined. All of the watersheds in our study have been logged in the past, and 11 watersheds currently have logged or exposed areas. Slaney and Martin (1997) point out that past logging practices have seriously impacted the habitat capacity of streams. Compounding the problem on Vancouver Island is a natural, historic low-flow regime in most small streams (Water Survey of Canada 1975). Our analysis shows summer low-flow problems in 155 of 165 streams. Low flow resulted in our classifying spawning

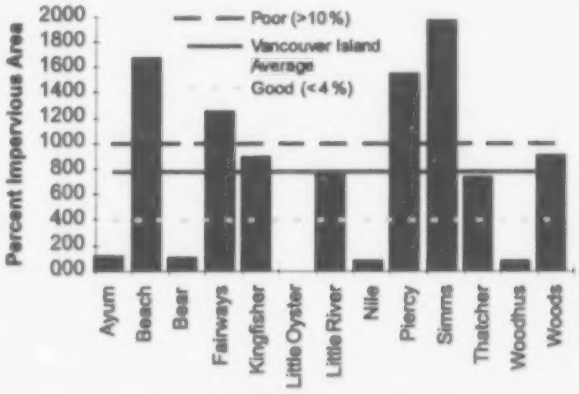


Figure 4. Percent Impervious Area for fourteen Vancouver Island watersheds.

Table 7. Type and percentage of each type of land use in 14 east coast Vancouver Island streams.

Watershed	% of each type of land use					
	Forest	Urban	Exposed or logged	Agriculture	Golf course	Rural residential
Ayum Creek	98.5	1.5	0	0	0	0
Beach Creek	58.1	22.1	7.8	7.4	4.6	0
Bear Creek	59.7	1.3	3.7	19.3	0	16
Fairways Creek	58.3	16.5	5.9	16	3.3	0
Kingfisher Creek	56.6	11.7	8.2	0	7.1	16.4
Little River and Scales Creek	54.9	10.2	9.7	22.3	2.8	0
Little Oyster River	98.8	0	1.2	0	0	0
Nile Creek	88.9	1.2	9.9	0	0	0
Piercy Creek	42.4	20.2	5	32.4	0	0
Simms Creek	60.8	25.8	13.4	0	0	0
Thatcher Creek	52.4	9.6	2.9	32.7	2.4	0
Woodhus Creek	99.0	1	00	0	0	0
Woods Creek	63.9	11.9	16.7	0	7.5	8
AVERAGE	68.6	10.2	6.5	10	2.1	3.1

and rearing habitat as severely degraded in 5 of 7 streams. We conclude that most of the 600 small streams on the east coast of the Island have similar water quantity problems. The consequences of long-term forest development and a natural low-flow regime is that small streams on Vancouver Island have reduced habitat capacity, making them particularly vulnerable to any development within their watersheds.

All sustainability problems are population driven. Urban and agricultural development have among the most extensive impacts on aquatic habitats in North America (Schueler 1991, Mitchell 1996). These land uses can affect flow during wet and dry periods, and can result in stream channelization, reduction in instream complexity, increases in sediment inputs, riparian alterations, water withdrawals for irrigation, compaction of substrate, and chemical inputs (Leopold 1968, Anderson 1970, Brown 1983, Klein 1990). Schueler (1994) examined a number of streams in urbanized areas of King County, Washington and found: they lacked significant pools; were often devoid of large woody debris; the riparian zones were often badly eroded; the bed gravels were generally packed with silts and sand; and the pattern of seasonal high and low flows were more frequent and longer lasting. Mitchell (1996) found that an average of 5 tons of sediment, organic matter, and bacteria per acre per year flow from crop lands to

Table 8. Percentage of impervious area (IA) based on the amount of roof tops and roads contained in land classified as urban in 14 east coast Vancouver Island watersheds.

Watershed	Watershed area (km^2)	Urban area (km^2)	IA due to roads[a]	IA due to roofs[b]	Total IA ($I_{road} + I_{roof}$)	% of watershed in IA[c]
Ayum Creek	13.7	0.21	0.13	0.03	0.16	1.17
Beach Creek	6.3	1.38	0.83	0.22	1.05	16.67
Bear Creek	8.3	0.11	0.07	0.02	0.09	1.08
Fairways Creek	3.6	0.59	0.35	0.1	0.45	12.5
Kingfisher Creek	2.8	0.33	0.20	0.05	0.25	8.93
Little River and Scales Creek	19.3	1.97	1.18	0.32	1.5	7.77
Little Oyster River	42	0	0	0	0	0
Nile Creek	16.9	0.19	0.11	0.03	0.14	0.83
Piercy Creek	7.7	1.56	0.94	0.25	1.19	15.45
Simms Creek	13.8	3.56	2.14	0.57	2.71	19.64
Thatcher Creek	4.56	0.44	0.26	0.07	0.33	7.28
Woodhus Creek	37.1	0.37	0.22	0.06	0.28	0.75
Woods Creek	11	1.31	0.79	0.21	1	9.09
AVERAGE	14.39	0.92	0.56	0.14	0.70	7.78

[a] I_{road} = 60% of Urban Area.
[b] I_{roof} = 40% x Urban Area - I_{road}.
[c] Impervious Area/Watershed Area.

Table 9. Percentage of impervious cover, status of stream quality, and ratings for 14 east coast Vancouver Island watersheds.

Watershed	Impervious area (%)	Status of stream quality	Rating
Ayum Creek	1.17	Minimum Impact	1
Beach Creek	16.67	Impacted	3
Bear Creek	1.08	Minimum Impact	1
Fairways Creek	12.5	Impacted	3
Kingfisher Creek	8.93	Stressed	1
Little River and Scales Creek	7.77	Stressed	1
Little Oyster River	0	Minimum Impact	1
Nile Creek	0.83	Minimum Impact	1
Piercy Creek	15.45	Impacted	3
Simms Creek	19.64	Impacted	3
Thatcher Creek	7.28	Stressed	1
Woodhus Creek	0.75	Minimum Impact	1
Woods Creek	9.09	Stressed	1

aquatic areas. He points out that besides choking aquatic life, these sediments deposit traces of fertilizer, herbicides, and insecticides to the recipient waterways.

Eighty-two percent of the watersheds we examined contained some level of urban development, and 43% of the watersheds had both urban and agricultural land use. Twenty-eight percent of our study creeks were impacted due to impervious surface area >10% and a further 28% were approaching that level. We found that streams had been simplified, resulting in reduced pool areas, a lack of LWD, and reduced instream cover. All but 1 stream had excessive fine sediments (Table 10). The population on the east coast of Vancouver Island is expected to double over the next 20 years. The question is, how do we protect the fish habitat we have left, and can we restore the habitats that are already degraded?

Protection of Small Stream Habitats

Loss of freshwater fish habitat cannot be compensated by quick fixes such as hatchery introductions or localized restoration efforts. We must implement a much more comprehensive approach that focuses on long-term planning, protection, restoration, and public awareness if we want to protect and restore our small stream habitat over the long term. We suggest: 1) planning initiatives that involve local government and local citizens; 2) development restrictions focused on long-term protection and attention to impervious area; 3) new legislation that restricts water extraction when flows are critical; 4) local government responsibility for habitat protection; 5) stronger, mandatory land development guidelines; 6) long-term riparian and instream restoration plans; and 7) public awareness and landowner contact programs.

We suggest that stewardship groups become involved in watershed planning through participation in the development of Official Community Plans (OCPs). We also recommend that volunteers form partnerships with local governments and become involved in the long-term planning of their watersheds by completing watershed management plans. These plans can provide community supported protection and development goals that ultimately become articulated in the OCP.

Development restrictions will help in the long-term protection of small stream habitats. These can be defined through stewardship bylaws that are environmentally focused, yet simple and administratively efficient, directions for small parcels of land (B.C. MELP 1996*b*). The Municipal Act contains provisions for: 1) tree protection/soil removal

Table 10. Status of fish habitat of 14 Vancouver Island watersheds.

Watershed	Percent pool area (<55%)	Large woody debris (<2)	Percent instream cover (<20%)	Percent fines (10–20%)	Percent wetted area (<90%)	Critical flow (<10% MAD)	Impervious surface area(>10%)
Ayum Creek	x				x	x	
Beach Creek		x		x	x	x	x
Bear Creek		x	x	no data	x	x	
Fairways Creek		x	x	no data	x	no data	x
Kingfisher Creek	x	x	x	x	x	no data	
Little Oyster R.		x	x	no data	x	x	
Little River		x		x	x	x	
Nile Creek		x	x	no data	x		
Piercy Creek	x	x	x	x	x	no data	x
Scales Creek	x	x		x	x	no data	no data
Simms Creek	x	x	no data	no data	x	no data	x
Thatcher Creek		x	x	x	no data	no data	
Woodhus Creek	x	x	x	no data	x	x	
Woods Creek		x		x	x	no data	

and deposition/watercourse protection bylaws; 2) zoning bylaws to protect sensitive areas; 3) development variance permits to protect site-specific areas; 4) development permits to protect lands in a natural state, particularly around watercourses; 5) subdivision bylaws to cover protection during construction; and 6) subdivision approval bylaws, which require an approving officer to enforce bylaws.

Impervious area represents the imprint of land development on the landscape (Schueler 1994). Severe changes in hydrology and channel morphology, and a significant and possibly irreversible loss of stream habitat and aquatic system function occurs once a watershed is covered by >10% impervious area (Booth and Reinelt 1993, Precision Identification Biological Consultants 1997, Green Mountain Institute 1998). We recommend that development planning ensure the amount of impervious area be kept to a minimum. Open space plans should be a mandatory part of each development, including provision of stormwater detention areas. These plans should require the developer to preserve an additional percentage of open space to accommodate the residents' future requirements for parks, playgrounds, and other community needs (Schueler et al. 1992). These types of development restrictions are critical if we want to reduce and control the amount of impervious area in a watershed.

The British Columbia Ministry of Environment, Lands and Parks (MELP) recognizes that low flow is a problem on Vancouver Island (B.C. MELP 1996*a*). It is now the policy of the Regional Water Management Branch that, when the natural mean monthly flow falls below 10% MAD, extractive licensed demands will only be allowed for the period when the mean monthly flow is above 60% MAD. This policy has resulted in 1 regional government developing headwater storage for their domestic water requirements and enhancing streamflows at the same time.

In her examination of the Fish Protection Act (FPA), Taves (1998) points out that section 5 of the legislation, which enables the protection of water flows for fish, is entirely discretionary. The Water Manager may or may not consider the impact on fish and fish habitat when making water licensing decisions. This legislation will amount to little, if any, protection for fish. We suggest that a better way to ensure flows for fish would be to elevate the Vancouver Island Regional Water Management policy to that of enforceable legislation. This would ensure flows for fish are recognized as equal to the needs of an expanding human population.

The greatest sediment loads to streams are exported during the construction phase of development (Schueler et al. 1992). It is the regional districts and municipalities that are responsible for approving land developments within their containment areas. However, it is the senior governments that are responsible for protecting habitat during and after development. We recommend that this function become the responsibility of the local governments. This would allow for a collaborative approach between the community stewardship groups and the local council throughout the development approval and construction phases. Local involvement will empower local citizens to protect their streams before any development is approved, and during development. It would also involve citizens in mitigation and restoration projects, which may accompany development. Finally, we contend that federal and provincial agencies should be responsible for enforcing environmental legislation. This would, in effect, make the enforcement body independent of the body approving the land use and development.

Our data shows that current environmental site planning techniques are not adequate to protect streams from the impacts of sedimentation. We recommend that regional districts and municipalities hire technical habitat protection staff to review and prescribe site-specific protection measures to accompany development plans. The cost for this staff should be covered, at least in part, by the developer, but the approval for the protection measures should be through the community. This would again vest the responsibility for environmentally responsible development in the hands of the community.

Currently, a number of management guidelines that apply to development practices are weak and applied inconsistently. Some of these guidelines focus on erosion control, revegetation, and leave-strip widths. For example, leave strips may or may not be required and their widths are currently discretionary and usually <30 m (Chilibeck et al. 1992). We contend that land development guidelines including leave strips must be reviewed and become mandatory. In Washington, researchers recently reviewed and revised their standard Riparian Habitat Area (RHA) widths to between 46 m for perennial or intermittent streams and 76 m for other streams and state shorelines (Knutson and Naef 1997). We must be equally diligent if we want to maintain riparian and small stream habitat structure and function.

Restoration of Small Streams

Loss of fish habitat can not be repaired by engineered structures alone (Booth and Reinelt 1993). This was exemplified in Oregon when the United States Forest Service attempted to restore anadromous salmon and steelhead runs in Fish Creek by installing instream structures to address a lack of pools, only to have the structures wash out during floods (Frissel 1997). Restoration efforts that focused on revegetating harvested slopes and restoring natural hydrographic regimes would have had more long-term benefits.

The long-term debate over the success of installing instream structures continues. Hartman and Miles (1995) found, on average, a 50–55% success rate on structures installed in the early 1980s in British Columbia. Since that time, however, acceleration of stream restoration projects in Oregon, Washington, and British Columbia has resulted in

increased knowledge about stream rehabilitation and project success has increased (House et al. 1991, Crispin et al. 1993, Heller et al. 1996, Cederholm et al. 1997). We believe that volunteers can implement successful restoration projects with the technical assistance of senior governments. We recommend that groups undertake habitat pre-assessments to determine what instream and riparian habitat factors are limiting and the appropriate priority for restoring these factors.

The effects of suspended sediment on rearing salmonids are significant. Sediment affects both the water column and the streambed (Griffith 1980). Light penetration to the streambed diminishes with increasing sedimentation, which, in turn, limits primary production and availability of food. Thompson (1972) reported decreasing potential productivity with decreasing substrate particle size, silt being the least productive. Salmonids are sight feeders and Bachman (1958) observed the cessation of feeding by cutthroat at turbidity levels of 25 ppm. Herbert and Merkens (1961) report that sediment levels as low as 90 ppm can adversely affect the survival of trout. Perhaps the greatest impact of sediment is on eggs and emerging fry. Bjornn (1969) and McCudden (1977) showed that the percentage of fine sediment in the gravels prevented the emergence of chinook and steelhead fry. When sediments reached 20–25% fines, emergence dropped from about 80% to <10%.

Sediment deposition is a natural process that takes place during periodic flooding, but accelerated upland erosion can increase sediment deposition in streamside areas because of downslope movement of dislodged soil. Riparian zones exert a direct biological, physical, and chemical influence on the stream (Connin 1991). A well-maintained riparian area may fill with sediment, but its eventual release is slow and gradual, and the damage to the stream habitat is minimal (Ruffin 1998). We suggest that riparian zones must be a priority for restoration if we want to minimize sediment inputs from urban and agricultural development. Eroding and altered stream banks must be replanted to ensure that sediment recruitment is minimized. Bioengineering solutions can be adopted in many soil stabilization and erosion control situations. Advantages of bioengineering include: low initial cost and lower long-term maintenance cost; low maintenance of live plants; environmental benefits to wildlife habitat; water quality improvement; improved aesthetics; improved strength over time as root systems develop; and compatibility with environmentally sensitive sites or sites with limited access.

We recommend that stewardship groups complete detailed riparian assessments of their watersheds when they find that the percentage of sediment in their streams exceeds 10%. These assessments should: 1) identify all sediment sources; 2) prioritize stream bank locations for restoration; and 3) suggest appropriate, preferably bioengineered, solutions. Once a riparian restoration assessment is in place, the stewardship group can develop a riparian restoration plan defining long- and short-term goals. The plan must include long-term riparian zone monitoring to ensure the integrity and maintenance of this critical component of fish habitat.

Forty-three percent of the 14 Vancouver Island streams we examined were pool-limited. Pools are a preferred macrohabitat type for coho juveniles in British Columbia (Glova and Mason 1976, Hartman 1965). Griffith (1980) investigated the summer habitat requirements of sympatric populations of cutthroat and coho in Bush, Holland, and Ayum creeks and found that the highest biomass usually occurred in pools. Nickelsen and Reisenbichler (1977) showed that 80% of the variation in coho standing crop in some Oregon streams was explained by pool area. Michalski and Reid (1998) found coho preferred artificial pool habitat with depths >30 cm.

There are a variety of ways to construct pool habitats, but volunteers must take care when designing a pool creation project (Cederholm et al. 1997, Slaney et al. 1997, Allan and Lowe 1997, Newbury et al. 1997). We do not recommend digging instream pools with a machine because this destabilizes the stream bed. Pools constructed this way also usually fill in. Over the past year, we have monitored machine-dug pools and found that 75% of them have filled in and they are causing erosion in a number of locations. We recommend that volunteers consult professionals, including senior government technical staff, for advice before any instream work is contemplated.

Instream cover provides fish with refuge from predators. It also increases the carrying capacity of streams by increasing habitat complexity. Prime cover for coho includes overhanging vegetation, undercut banks, rootwads, and log accumulations (Griffith 1980). Wesche and Rechard (1980) described trout habitat as, among other things, object oriented. Bustard and Narver (1975) showed that as water temperatures dropped to 47–35°F, juvenile coho and steelhead moved into deeper pools containing upturned tree roots and logs. Research in Oregon shows that the addition of conifer logs to debris-poor streams can increase salmonid smolt production several fold (Murphy 1995). The addition of debris in the Clearwater River, Washington produced a 5-fold increase in winter survival of juvenile coho (Cederholm et al. 1988).

In small and intermediate streams, LWD contributes to channel stabilization, energy dissipation, and sediment storage (Cederholm et al. 1997). Bilby and Ward (1991) found a positive correlation between the pool area of a stream and the volume of the LWD forming the pool. LWD promotes the storage of sediment and traps fish carcasses, which provide a source of nutrients and carbon (Slaney et al. 1997). Peters et al. (1992) found that introduced woody debris is at least as effective as natural debris in attracting and sustaining

juvenile coho salmon during summer. Enhancing LWD can be done at relatively low cost; however, we recommend that volunteers have expert advice to ensure that LWD placements and log sizes are appropriate.

Finally we suggest that volunteers augment restoration projects with public information programs. Many stewardship groups working on stream assessment and restoration activities have been confronted with a general lack of knowledge on the part of watershed residents about the importance and sensitivity of streams (Shepp and Cummins 1997). Often the general public does not understand their connection to small streams and the associated ecosystem. Yet the understanding and support of watershed residents is critical to the success of long-term restoration and protection efforts We suggest that volunteers host neighbourhood information sessions and/or landowner contact programs before and during restoration projects. Information on the project, as well as on the importance of riparian and instream habitat, should be emphasized. Discussions of concerns about fish habitat protection, including private land stewardship, should also be encouraged and facilitated.

Increasing the public's knowledge and appreciation about the importance and fragility of small stream habitats may well be the most important activity that community stewards can undertake. It may also tip the scales from continued deterioration to long-term protection and restoration of east coast Vancouver Island streams.

ACKNOWLEDGEMENTS

We sincerely thank the stewardship groups that conducted USHP fish habitat assessments on their creeks. The dedication of these volunteers to identifying the factors limiting fish production has lead not only to comprehensive, prioritized restoration plans, but also to a much better understanding of the status of fish habitat on Vancouver Island.

We also thank L. Boykiw and D. Hart for their diligence during the data analysis, P. Wipper for his advice regarding impervious surface area in Nanaimo, and D. Fetherston for his assistance in the preparation of the figures for this report.

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Forest Management for Biodiversity in Montane "Islands"

Joan Voller
British Columbia Ministry of Forests, Research Branch
712 Yates Street, Victoria, BC, V8W 3E7, Canada

Andy MacKinnon
British Columbia Ministry of Forests, Research Branch
712 Yates Street, Victoria, BC V8W 3E7, Canada
Andy.MacKinnon@gems1.gov.bc.ca

ABSTRACT

The montane is often considered an "island" habitat surrounded by different and sometimes inhospitable ecosystems, where factors such as area, isolation, and immigration influence species richness and diversity. Until recently, the montane has provided only a small portion (2–3%) of the forest harvest in coastal British Columbia. Montane forests were considered to have relatively inoperable terrain, with wood that had undesirable fibre quality and moderately low productivity. However, as submontane ecosystems are logged, development plans call for timber supply to come increasingly from coastal montane forests. Unfortunately, our knowledge of coastal montane ecosystems is meagre, and most current management practices employed there are based on research and management experience in the submontane. Further, montane ecosystems are more sensitive to disturbance and slower to recover than submontane ecosystems. In order to responsibly manage these habitats we have to understand the differences between montane ecosystems and the lower elevations for which the current management strategies were created. By understanding the species present in the montane, and their habitat associations, we can develop more appropriate management strategies. So far our project has focused mainly on a general inventory of plants, habitat elements, vertebrates, and arthropods found at our site on northern Vancouver Island. We are using our data to investigate species/habitat associations. This information can then be used to better understand habitat needs, and thus develop more appropriate management strategies for both the listed and common species found during our inventory.

Key words: forest management, montane, Vancouver Island.

Coastal montane forests are the transition zone between low (submontane) and high (subalpine) elevation forests, and range from approximately 650 to 1,000 m. The montane and subalpine elevations offer habitat for a large range of plants and animals, as well as providing economic value from recreation and resource extraction such as forest harvesting.

Concern for the preservation of the montane and subalpine forests has increased with the elevated harvest of these areas. Until recently, the montane has contributed only a small portion (2–3%) of the forest harvest in coastal British Columbia. However, the montane and subalpine forests of Vancouver Island actually account for about 30% of the land base (Smith 1992) and represent, for example, 25% of MacMillan Bloedel's future Annual Allowable Cut (Beese 1995). In the past, these areas were considered to have relatively inoperable terrain with wood that had undesirable fibre quality and moderately low productivity. However, as submontane ecosystems are logged, development plans call for timber supply to come increasingly from coastal montane forests. Unfortunately, our knowledge of coastal montane ecosystems is meagre, and most current management practices employed here are based on research and management experience from the submontane.

In order to develop more appropriate management strategies for the montane and subalpine forests we need to understand the differences between these areas and the submontane. Montane ecosystems are more sensitive to disturbance and slower to recover than submontane ecosystems. The montane's greater sensitivity to disturbance is partially due to its harsher climate. The harsher climate of the higher elevations produces lower temperatures, greater snowpacks, greater amounts of annual precipitation, and generally higher winds. This harsher climate also reduces the growing season and thus raises concerns over the survival, growth, and time needed to regenerate the forests (e.g., Koppenaal and Mitchell [1992] for montane forests, and Brett and Klinka [1998] for subalpine forests). As well, the greater snowpacks reduce the time that food and other necessities are available to wildlife, which influences species richness and diversity.

Unlike the submontane, the montane and higher elevations act like "island" habitats surrounded by different and sometimes inhospitable ecosystems. Species richness and diversity are affected by factors such as area, isolation, and

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 498pp.

immigration. Because the montane is often considered an "island," a species' ability to survive will greatly depend on the degree to which it has adapted to these factors. Lomolino and Davis (1997) describe montane islands of North America as "complex systems, isolated by a heterogeneous mix of immigration filters." How isolated these systems are for a species depends mainly on the habitat that lies between these "islands" and a species' ability to cross it. This ability to cross an inhospitable habitat will affect a population's potential to avoid local extinction. A certain amount of immigration must occur in order to limit the risk of local extinction (i.e., immigration rate must exceed extinction rate; MacArthur and Wilson 1967, Lomolino and Davis 1997). The degree of isolation between montane islands also depends somewhat on distance, and a species' ability to cover that distance. However, artificially decreasing the amount of isolation between populations can also cause problems for a species (e.g., the Vancouver Island marmot's range has become concentrated by their using clearcuts instead of travelling to the nearest mountain meadow, thus making the whole population more susceptible to mortality factors).

The Coastal Montane Biodiversity (CMB) project on Mount Cain was initiated to help increase our understanding of the montane and subalpine forests of coastal British Columbia. By understanding the species present in the montane and their habitat associations, we can develop more appropriate management strategies. The data presented in this paper are a small subset of data collected so far on this project. The inventory data collected during this project are in the process of being analyzed. In addition, other research projects associated with the CMB project are focusing on cavity nesting birds, bats, and ground and arboreal arthropods of the montane (Joy 2000, Kellner 1999, Fagan and Winchester 1999).

STUDY AREA

The study area is located at Mount Cain (50°3'N, 126°18'W) and Maquilla Peak near the community of Woss, B.C. on the northern part of Vancouver Island. Mount Cain provides the headwaters of the Tsitika River and, through Cain Creek, drains into the Davie River, which is the major drainage separating Mount Cain from Maquilla. The Englewood Logging Division of Canadian Forest Products, Ltd. (CANFOR) manages the area as part of Tree Farm License No. 37.

The landscape consists of a matrix of managed and natural forest stands that are part of 2 biogeoclimatic zones (Meidinger and Pojar 1991). The first and largest portion of the study area is classified as the Montane, Very Wet Maritime Coastal Western Hemlock variant (CWHvm2). This variant occupies an elevation range of about 650–1,000 m (600–900 m at Mount Cain) on the coastal mainland of British Columbia, the Queen Charlotte Islands, and Vancouver Island (Meidinger and Pojar 1991). Cool, short summers, cool winters, high precipitation (mean annual precipitation of 2,787 mm) and substantial snowfall are some of the climatic characteristics of the CWHvm2 variant (Meidinger and Pojar 1991).

The second biogeoclimatic unit in the study area is the Windward, Moist Maritime Mountain Hemlock variant (MHmm1). This variant covers the subalpine regions of coastal British Columbia, Vancouver Island, and the Queen Charlotte Islands, with lower limit elevations between 800 and 1,000 m and upper limits between 1,100 and 1,350 m (Green and Klinka 1994). At Mount Cain this variant occurs between 900 and 1,300 m. The variant is characterized by short, cool, moist summers and long, wet, cold winters with deep accumulations of snow (total mean annual depth of approximately 820 cm; Meidinger and Pojar 1991). The Parkland subzone of the MHmm1 occurs at the upper elevations (MHmm1p), with the nonforested ecosystems of the Alpine Tundra (AT) above that.

At lower elevations, below the study area, the Submontane Very Wet Maritime Coastal Western Hemlock variant of the CWHvm (CWHvm1) occurs.

The CWHvm2 variant supports western hemlock (*Tsuga heterophylla*) and western redcedar (*Thuja plicata*), interspersed with varying amounts of amabilis fir (*Abies amabilis*). Some drier sites at lower elevations also support small amounts of Douglas-fir (*Pseudotsuga menziesii*), while yellow-cedar (*Chamaecyparis nootkatensis*) is present at higher, moister regions of the zone. Major understory species include Alaskan blueberry (*Vaccinium alaskaense*), five-leaf bramble (*Rubus pedatus*), step moss (*Hylocomium splendens*), lanky moss (*Rhytidiadelphus loreus*), and pipecleaner moss (*Rhytidiopsis robusta*; Green and Klinka 1994).

Mountain hemlock (*Tsuga mertensiana*), yellow-cedar, and amabilis fir dominate the subalpine MHmm1 variant. Both western hemlock and western redcedar are also present in this zone, though at lower elevations. The forest becomes increasingly sparse with increasing altitude and thins into parkland with trees scattered in isolated and irregular clumps (Meidinger and Pojar 1991). As with the CWHvm2 variant, dominant vegetation of this variant includes Alaskan blueberry and pipecleaner moss, in addition to oval-leafed blueberry (*Vaccinium ovalifolium*).

METHODS

The inventory of biodiversity focused on 4 major life-form groups: plants, habitat elements, vertebrates, and arboreal- and ground-associated arthropods. The compilation of data on the invertebrates found during our inventory is unfinished; therefore, they are not listed in this paper (Fagan and Winchester 1999). Whenever possible, inventory techniques were those established by the Resource Inventory

Committee (RIC). Climate data were also collected.

Fifty 750-m-long transects were established through a representation of ecosystem types. Five of the 50 transects were established along an elevational gradient from 650 m to 1200 m. Six square sample plots (20 m^2, 0.04 ha) were established at 150-m intervals along each of the 50 transects. In order to increase independence (>300 m apart), plots 1, 3, and 5, of each transect were established as "biodiversity" plots.

Details of methods used for the inventory may be found in the CMB final report (Joy et al. 1999). Details of methods for the associated research projects may be found in the following: cavity nesting birds (Joy 2000); bats (Kellner 1999); arthropods (Fagan and Winchester 1999).

RESULTS

The species that are most likely to be affected by harvesting in the montane are rare species and species dependent on the higher elevation forests for part or all of their life cycle. Table 1 shows the species at risk that were found during our inventory or that are known to occur in the montane on Vancouver Island. By including those species that are known to occur in the montane but were not found during our inventory, we recognize that either their rarity, or a gap in our methods, may have prevented us from finding them. Through our inventory of the Vancouver Island montane and a Rare Element Occurrence Report compiled by the British Columbia Conservation Data Centre (CDC; CDC 1998), we have noted 3 bryophyte and lichen, 40 vascular plant, 7 bird, and 6 mammal species at risk. Of these species at risk, 14 vascular plant, 3 bird, and 1 mammal species are dependent on the higher elevations for part or all of their life cycles. The following additional bird species that are not at risk, but that are dependent on the higher elevations for part or all of their life cycle, were also found during our inventory: three-toed woodpecker (*Picoides tridactylus*), gray jay (*Perisoreus canadensis*), American pipit (*Anthus rubescens*), and gray-crowned rosy finch (*Leucosticte tephrocotis*).

In addition, we found 321 common vascular plant species, 71 common bird species, and 9 common mammal species during our inventory of Mount Cain; for more specific results on the common species found during the inventory refer to Joy et al. (1999). For results from the associated research projects refer to Joy (2000) for cavity nesting birds, Kellner (1999) for bats, and Fagan and Winchester (1999) for arthropods.

DISCUSSION

Increased concern for the preservation of the higher elevations has initiated an increase in interest in alternative silvicultural systems (e.g., shelterwood, green tree retention, and patch cuts). Projects such as Montane Alternative Silvicultural Systems (MASS; Arnott et al. 1995) and Sicamous Creek (Hollstedt and Vyse 1997) have given us the opportunity to study the potential of these alternatives, both from an economic viewpoint and a biological viewpoint.

The Sicamous Creek project also looks at the effects alternative silvicultural systems have on biodiversity. No coastal project yet addresses these concerns. However, MASS characterizes forest structure and natural vegetation dynamics following a variety of silvicultural treatments (Beese et al. 1995). Combining the lessons of MASS with the lessons from the CMB project on the habitat requirements of various montane forest species, we can evaluate the efficacy of different silvicultural systems in meeting habitat requirements for montane forest species.

For example, Beese et al. (1995) reported that the pre-logging basal area that remained after logging was 18% (206 stems/ha >17.5 cm dbh) for the shelterwood treatment, and 5% (22 stems/ha >17.5 cm dbh) for the green tree retention treatment. On Mount Cain, Joy (2000) found that nest tree characteristics for the red-breasted sapsucker were at least 32.5 m in height and 93.3 cm in diameter, across a range of decay classes. He also found that the red-breasted sapsucker preferred western white pine (*Pinus monticola*), amabalis fir, western hemlock, and mountain hemlock, and suggests that these species should be included in the design of wildlife tree patches. It is widely felt that primary cavity nesters are keystone species in northern forests (Daily et al. 1993, Walters 1996). Joy (2000) therefore suggests that wildlife tree retention in coastal montane forests be based on mean nest tree diameter and height. Combining the findings from these studies will help us meet the habitat requirements of primary cavity nesters such as the red-breasted sapsucker; at the same time we also benefit other species that use the cavities created by these species.

In order to develop effective alternative silvicultural systems for the montane and subalpine forests we also have to understand how the "island" effect may influence species populations. Coastal montane ecosystems are not subject to regular, large-scale, stand-destroying disturbances (e.g., Beese et al. 1995), and so might be expected to harbour some species with poor dispersal abilities. For example, Winchester (1997) reports an unusually high incidence of wingless arthropods in canopies of coastal temperate rain forests on western Vancouver Island. Similar results are anticipated from canopy arthropod work at Mount Cain.

Vertebrates are generally less affected by the isolation factor. Of the 13 listed and 84 common vertebrate species in the Vancouver Island montane, all but 4 (three-toed woodpecker, American pipit, white-tailed ptarmigan, and Vancouver Island marmot) can be found at all elevations throughout the year, and are not dependent on the montane and higher elevations for any of their life cycle; therefore, isolation is not a concern. As well, these 4 vertebrate species that live at least part of their life cycle in the montane are capable of

Table 1. Species at risk that have been noted in the montane on Vancouver Island.

Species	Status[a]	Source[b]	Dependence[c]
Bryophytes and Lichens			
Bryoria nadvornikiana	?	Mt. Cain	
Chaenotheca phaeocephala	?	Mt. Cain	
Pseudocyphellaria rainierensis	?	Mt. Cain	
Vascular Plants			
Allium cremulatum Olympic onion	Red S1 G4	Other	
Anemone drummondii var. *drummondii* Drummond's anemone	Blue S2S3 G4T4	Other	Y
Apocynum sibiricum var. *salignum* Clasping-leaved dogbane	Blue S1? G5?T?	Other	
Artemisia furcata var. *heterophylla* Three-forked mugwort	Blue S1? G4T?	Other	Y
Asplenium adulterinum Corrupt spleenwort	Blue S1? GU	Other	Y
Aster paucicapitatus Olympic mountain aster	Blue S2S3 G3?	Other	
Botrychium simplex Least moonwort	Blue S1? G5	Other	
Carex feta Green-sheathed sedge	Blue S2S3 G5	Other	
Disporum smithii Smith's fairybells	Blue S2S3 G5	Other	
Douglasia laevigata Smooth douglasia	Blue S2S3 G3?	Other	
Draba lonchocarpa var. *vestita* Lance-fruited draba	Blue S2S3 G4T3	Other	Y
Dryopteris arguta Coastal wood fern	Blue S2S3 G5	Other	
Enemion savilei Queen Charlotte isopyrum	Blue S3 G3	Other	
Epilobium ciliatum ssp. *watsonii* Purple-leaved willowherb	Blue S2S3 G5T?	Other	
Epilobium glaberrimum ssp. *fastigiatum* Smooth willowherb	Blue S2S3 G5T?	Other	Y
Epilobium leptocarpum Small-flowered willowherb	Blue S2S3 G5	Other	Y
Erysimum arenicola var. *torulosum* Sand-dwelling wallflower	Blue S1? G4G5T?	Other	Y
Erythronium montanum white glacier lily	Blue S2S3 G4	Other	Y
Euphorbia serpyllifolia Thyme-leaved spurge	Blue S2S3 G5	Other	
Galium trifidum ssp. *trifidum* Small bedstraw	Blue S1? G5T?	Other	
Geum schofieldii Queen Charlotte avens	Red S2 G2Q	Other	
Githopsis specularioides Common bluecup	Red S2 G5	Other	
Hedysarum occidentale Western hedysarum	Blue S2S3 G5	Other	Y
Hypericum scouleri ssp *nortoniae* Western St John's-wort	Blue S2S3 G5T?	Other	
Lewisia triphylla Three-leaved lewisia	Blue S1? G4?	Other	Y
Lophochlaena refracta var. *refracta* Nodding semaphore grass	Blue S2S3 G4G5T4	Other	
Myriophyllum quitcense Waterwort water-milfoil	Blue S2S3 G4?	Other	
Nothochelone memorosa Woodland penstemon	Blue S2S3 G5	Other	
Orthocarpus imbricatus Mountain owl-clover	Red S1 G5	Other	Y
Pyrola elliptica White wintergreen	Blue S1? G5	Other	Y
Rubus lasiococcus Dwarf bramble	Blue S2S3 G5	Other	Y
Rubus nivalis Snow dewberry	Blue S2S3 G4?	Other	Y
Rupertia physodes California-tea	Blue S2S3 G4	Other	
Sanguisorba menziesii Menzies' burnet	Blue S2S3 G3G4	Other	
Senecio macounii Macoun's groundsel	Blue S2S3 G5	Other	
Thelypteris nevadensis Nevada marsh fern	Red S1 G4	Other	
Viola biflora ssp. *carlottae* Queen Charlotte twinflower violet	Blue S3 G5T3	Other	
Viola howellii Howell's violet	Blue S2S3 G4	Other	
Yabea microcarpa California hedge-parsley	Blue S1? G5?	Other	
Birds			
Marbled Murrelet *Brachyramphus marmoratus*	Red S2B, SZN G3G4	Mt. Cain	
Northern Goshawk *Accipiter gentilis laingi*	Red S2B G5T2	Mt. Cain	
Northern Pygmy Owl *Glaucidium gnoma swarthi*	Blue S3 G5T3Q	Mt. Cain	Y
Peale's Peregrine Falcon *Falco peregrinus pealei*[d]	Blue S3B, SZN G4T3	Other?	
American Peregrine Falcon *F. p. anatum*[d]	Red S2B, SZN G4T3	Mt. Cain?	
Pine Grosbeak *Pinicola enucleator carlottae*	Blue S3S4B SZN G5T3T4	Mt. Cain	Y
Sandhill Crane *Grus canadensis* (this species only noted above the canopy)	Blue S3B,SZN G5	Mt. Cain	
White-tailed Ptarmigan *Lagopus leucurus saxatilus*	Blue S3 G5T3	Mt. Cain	Y

Table 1. Continued

Species	Status[a]	Source[b]	Dependence[c]
Mammals			
Keen's long-eared myotis *Myotis keenii*[e]	Red S1S3 G2G3	Mt. Cain?	
Roosevelt elk *Cervus elaphus roosevelti*	Blue S2S3 G5T4	Mt. Cain	
Vancouver Island ermine *Mustela erminea anguinae*	Blue S3 G5T3	Mt. Cain	
Vancouver Island marmot *Marmota vancouverensis*	Red S1 G1G2	Other	Y
Vancouver Island wolverine *Gulo gulo vancouverensis*	Red S1 G4T1	Other	
Water shrew *Sorex palustris brooksi*	Red S2 G5T2	Other	

[a] From British Columbia Conservation Data Centre (CDC).

[b] Mt. Cain = found during our inventory; Other = compiled from information on the CDC Tracking Lists (CDC 1999) and Rare Element Occurrence report for montane areas within the CWH and MH zones on Vancouver Island (CDC 1998).

[c] Y indicates species is dependent on higher elevations for all or part of its life cycle.

[d] A peregrine falcon was seen during our inventory, but we were unable to tell which subspecies it was.

[e] Due to the inability to distinguish this species from the western long-eared myotis we can not confirm that this species was present during our inventory.

dispersing the required distance between montane "islands."

As well, of the 40 listed plant species found in the Vancouver Island montane, 14 are found only in the montane. Like the vertebrates, these plant species are thought to be capable of dispersing the required distance between montane "islands" so isolation is not a concern.

The vertebrate and plant species that are dependent on the higher elevations for all or part of their life cycle are more likely to be affected by loss of area or loss of important habitat attributes. For example, Martin (1997) states that habitat loss due to forestry and recreational activities has the potential to impact summer production, winter survival, and dispersal of the white-tailed ptarmigan on Vancouver Island. As well, species such as the northern pygmy owl and the three-toed woodpecker rely on the availability of wildlife trees.

The upper elevations are also important habitat for the more common species. Ogle and Martin (1997) reported that in the alpine, avian species diversity (mostly migratory species) and food resources were high in the late summer and fall. At this same time, they observed that food resources, as well as avian diversity, were on the decline in lower elevation forests (Ogle and Martin 1997), thus noting the importance of higher elevations to migratory birds.

The increased demand on the resources of the montane ecosystem requires special attention. We need to deal with the increasing problem of managing biological diversity, as well as understanding how the principles of biodiversity management interact with the sustainability of all species, including endangered, rare, and regionally significant animals and plants.

ACKNOWLEDGEMENTS

We are grateful for financial, technical, administrative, and logistical support from the British Columbia Ministry of Forests, Research Branch; British Columbia Ministry of Environment, Lands and Parks, Region One; Canadian Forest Products Ltd., Englewood Logging Division; British Columbia Conservation Fund; and Forest Renewal British Columbia. We also thank the University of Victoria and Simon Fraser University for their cooperation and support. We also wish to thank all those involved in this project for their dedication and hard work.

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Nest and Nest Structure Preferences of Red-Breasted Sapsuckers in Coastal High Elevation Forests

J.B. Joy
598 Millstream Lake Rd, Victoria, BC, V9E 1E7, Canada
jeffrey_joy@bc.sympatico.ca

ABSTRACT

The red-breasted sapsucker (*Sphyrapicus ruber*) is the most abundant primary cavity excavator in coastal montane forests, and therefore a crucial component of the community. Red-breasted sapsuckers provide nesting and roosting opportunities for many species including the blue-listed northern pygmy owl (*Glaucidium gnoma*). In 1997 and 1998, as part of the Coastal Montane Biodiversity Project, I recorded the nest tree characteristics for 32 nests of the red-breasted sapsucker at Mt. Cain and Maquilla peak in the high elevation coastal forests of northern Vancouver Island. Nests were located in dead western white pine (*Pinus monticola*), mountain hemlock (*Tsuga mertensiana*), western hemlock (*Tsuga heterophylla*), and Douglas-fir (*Pseudotsuga menziesii*) trees. The mean diameter at breast height was significantly greater for nest trees than for random samples of dead trees. Nest trees were significantly taller than a random sample of dead trees. Nest height was significantly related to tree height. All nest structures discovered were in moderate decay stages. Red-breasted sapsuckers are selecting large diameter, moderately decayed, tall nest trees in these habitats. In placement of their nest cavities red-breasted sapsuckers are probably balancing predation risk (high nest heights) and adequate nest space (diameter) to choose an optimal location that maximizes reproductive output. The orientation of nest cavities did not differ from a random distribution. To maintain red-breasted sapsuckers and the suite of species that depend upon it in coastal high elevation forests, habitat managers must maintain tall large diameter trees that will develop appropriate internal decay conditions in wildlife tree patches.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Assessing Endangered Ecosystems in the Inland Skeena Region

Sybille Haeussler
Skeena Forestry Consultants
4279 Rue Boyer, Montreal, PQ, H2J 3C8, Canada
skeena@internet.uqam.ca

Anne Hetherington
British Columbia Ministry of Environment, Skeena Wildlife Branch
Bag 5000, Smithers, BC, V0J 2N0, Canada

ABSTRACT

The southeastern Skeena region of British Columbia supports a rich variety of ecosystems with distinctive transitional or ecotonal characteristics. A 2-year pilot study was carried out to inventory and assess rare and endangered terrestrial ecosystems in the low-elevation Sub-Boreal Spruce (SBS) and Interior Cedar–Hemlock (ICH) biogeoclimatic zones. Scrub-steppe or savanna-steppe, grassland, and herbaceous meadow ecosystems are highly threatened by changes to fire and grazing regimes, invasive species, and urban/rural development. Riparian or swamp forest ecosystems are other centres of local biodiversity at risk on private, municipal, and Crown land. The Red- and Blue-listing scheme for plant communities is an important first step in raising awareness about rare ecosystems, but will provide little protection in populated areas unless human communities adopt these ecosystems as symbols of local uniqueness and identity.

Key words: floodplain, grasslands, inventory, mapping, scrub-steppe, Skeena, swamp forest.

Many speakers at this conference (Eastman 2000, Pojar 2000, Scudder 2000) have called for a shift in conservation efforts from a species-centered to a whole-ecosystem approach. To reinforce that message we paraphrase Rowe (1994, 1998): It is time for biologists and ecologists to set aside the conviction that the entities of prime importance on earth are plants, animals, and, especially, people. Our focus must instead become "the globe's miraculous life-filled skin," acknowledging that life arises not within strands of DNA, but from the complex air-land-water support system that provides organized forms of matter and energy to all organisms.

Reed Noss, keynote speaker at this conference, outlined how conservation of regional ecosystems on a very large scale is needed to retain viable wildlife populations. However, as noted by our plenary speakers, ecosystems of highest value and diversity typically lie outside potential protection corridors in areas of high human population, where it is unrealistic to expect a complete cross-section of habitats to receive full or even partial protection. By default, the target of conservation efforts in such areas must often be those unusual, diverse, and highly productive ecosystems that are at highest risk. This paper reports on a pilot project to begin that process by identifying, mapping, and assessing endangered ecosystems in a portion of the Skeena region of British Columbia.

THE INLAND SKEENA REGION

The study area is located in west-central British Columbia, just east of the spine of the Coast Mountain Range. It lies mainly within the watershed of the Skeena River and its major inland tributaries (Bulkley, Babine, and Kispiox rivers), but spills over into the far reaches of the Fraser system, taking in the network of large lakes (Francois, Ootsa, etc.) at the western edge of the Nechako Plateau. Three ecoprovinces, 5 ecoregions, and 8 ecosections, as well as 7 biogeoclimatic zones, 10 biogeoclimatic subzones, and 6 biogeoclimatic variants are represented (Pojar et al. 1988, DeMarchi et al. 1990, Banner et al. 1993). It has a total area of 4.6 million ha, encompassing 4 forest districts (Bulkley, Kispiox, Lakes, and Morice), and taking in much of the traditional territories of the Gitksan, Wet'suwet'en, Nat'oot'en, Lake Babine, and Cheslatta, as well as peripheral territories of several other First Nations. The study area is not restricted by land ownership, tenure, or zoning, and includes the municipalities of Hazelton, Smithers, Telkwa, Houston, and Burns Lake.

In terms of endemism, species richness, or rarity, the inland Skeena region does not rank as a global biodiversity hot spot. But, like all regions of British Columbia, it has unique characteristics that contribute to the ecological diversity of this richly endowed province. Specifically, the study area occupies the junction of 3 biomes (the temperate rainforest, the taiga, and the [inter]montane forest-steppe), with many plant and animal species at northern, southern, or eastern

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

peripheries of their ranges, and has an exceptionally broad transition zone between coastal and interior climatic zones. Cannings and Cannings (1996) recognize the area as a biological suture zone, where biota formerly separated by glaciation come into contact. It is also situated at the northern margin of the intensely developed temperate urban-agricultural zone. The inland Skeena region thus supports a great variety of minimally damaged ecosystems, many with distinctive transitional or ecotonal characteristics.

INVENTORY METHODS

The 2-year inventory focused on Red-listed grassland, scrub-steppe, and floodplain ecosystems in the Bulkley Basin (BUB) ecosection and dry cool Sub-Boreal Spruce (SBSdk) biogeoclimatic subzone, with the objective of mapping all Red-listed occurrences in the Bulkley District SBSdk at 1:20,000 scale. Other objectives were to provide an overview assessment of rare and endangered ecosystems throughout the 4 forest districts, including plant communities not currently listed, and to raise awareness about rare and endangered ecosystems and their management, among British Columbia Ministry of Environment, Lands and Parks and Ministry of Forests staff in the 4 districts.

Site visits were made to 131 rare community element occurrences. Full or reconnaissance-level ecosystem descriptions (B.C. MOF and B.C. MELP 1996, Cadrin et al. 1996) were made at each site, and plant voucher specimens and rock samples were collected. Each ground-truthed element occurrence was marked on aerial photographs, and photographic slides were taken. Fifty-seven permanent photo-tie-points were established to allow changes in ecosystems to be monitored. Field records from 107 additional element occurrences were obtained from other sources.

Twenty-six 1:20,000 TRIM (Terrain Resource Information Management) mapsheets were either partially or completely mapped with 871 plant community polygons delineated (total area 6,795 ha). These mapsheets encompass the Bulkley Valley SBSdk corridor, with a few partial mapsheets completed in other drainages and in other forest districts. The project maps were digitized in ARC-INFO format by the British Columbia Conservation Data Centre (CDC) and linked to a computerized database that is compatible with Terrestrial Ecosystem Mapping (TEM) standards (Cadrin et al. 1996) and allows each element occurrence or polygon to be queried.

RARE AND ENDANGERED ECOSYSTEMS

Rare and endangered ecosystems usually possess 1 or more of the following features:

- unusual physiographic setting (climate, geology);
- unusual or exceptionally diverse biota (e.g., peripheral species);
- unusual or exceptionally complex disturbance history;
- attractive to humans.

These attributes are highly interrelated. The more attributes a particular ecosystem possesses, the more rare and endangered it is likely to be.

The CDC Red List currently includes 3 plant communities that fell within the purview of our study (Table 1). There are 2 situations where these Red-listed ecosystems and other rare-but-unlisted ecosystems most commonly occur: (a) warm, dry, southwest-facing slopes with mineral-rich parent materials; and (b) riparian or rich minerotrophic swamp systems. Both are low-elevation, nutrient-rich environments with active disturbance cycles (dominated in the first case by fire, and in the second case by flooding), heavy wildlife use, and high biotic diversity. They also have some of the highest agricultural capability in the region and a long history of use by both aboriginals and European settlers.

SOUTHWEST-FACING STEPPE, GRASSLAND, AND MEADOW ECOSYSTEMS

Saskatoon - Slender Wheatgrass Scrub Steppe

Saskatoon - slender wheatgrass is a very diverse xeric to submesic scrub-steppe community found in the SBSdk, SBSmc, and ICHmc subzones (Pojar et al. 1984, Banner et al. 1993). It is a mosaic of stunted saskatoon (*Amelanchier alnifolia*), snowberry (*Symphoricarpos albus*), cherry (*Prunus virginiana*, *P. pensylvanica*), and prickly rose (*Rosa acicularis*) scrub, mixed with patches of grasses (*Elymus* spp., *Stipa* spp., *Festuca saximontana*, *Poa* spp.) and forbs (peavine [*Lathyrus nevadensis*], northern bedstraw [*Galium boreale*], yarrow [*Achillea millefolium*], wild onion [*Allium cernuum*]). Thickets of taller willow (*Salix scouleriana*), aspen, cherry, and hawthorn (*Crataegus douglasii*) are common. With their high insolation and low snowpacks, these ecosystems provide critical habitat for many wildlife species, notably mule deer (*Odocoileus hemionus*) and western garter snakes (*Thamnophis sirtalis*), and support xerophytic plant and insect species not found elsewhere in the landscape.

In total, 187 occurrences of scrub-steppe (937 ha) were mapped in the Bulkley District. Seventy-four occurrences were recorded in other forest districts (Table 1), but casual inspection of aerial photos and satellite imagery suggests there are several hundred more occurrences. Although not particularly rare, this plant community is threatened by changes to its disturbance regime, including climatic change, fire suppression, urbanization, heavy grazing or browsing, and invasive nonnative plants. There is excellent

Table 1. Current and proposed Red- and Blue-listed plant community occurrences mapped in the inland Skeena region.

	BEC[a] site series	TEM[b] map code	current list	proposed list	Bulkley Forest District n	Bulkley Forest District ha	Bulkley Forest District %E/G[d]	Bulkley Forest District % Crown[e]	Other Districts[c] n
Plant Community									
Saskatoon - slender wheatgrass scrub steppe	SBSdk/81	SW	Red	Blue	226	1074	39	37	74
Bluegrass - slender wheatgrass grasslands	SBSdk/82	BW	Red	Red	5[f]	58	73	73	14
Black cottonwood - red osier dogwood - prickly rose floodplain forest	SBSdk/08	CD	Red	Red?	227	1363	36	22	40
Western redcedar - Roche spruce - horsetail - skunk cabbage swamp forest	ICHmc2/07	RC	Blue	Red?	1	22	100	100	no data
Douglas-fir - juniper - pinegrass open forest	SBSdk/04	DSxw	Blue	Red?	0				2
Rocky mountain juniper - saskatoon - slender wheatgrass savanna steppe	undescribed	JS	unlisted	Red	13	84	87	0	2
Aster - meadow rue - peavine - fireweed mesic montane meadow	undescribed	AM	unlisted	Red	54	148	51	77	13
Trembling aspen - wild cherry - beaked hazelnut deciduous scrub	ICHmc2/55	bt	unlisted	Red	0				5
Bracken - cow parsnip - fireweed fern meadow	undescribed	CA/br	unlisted	Red?	4	143	18	50	2
Hybrid white spruce - common horsetail floodplain forest	SBSdk/07a	SHa/e	unlisted	Red	36	108	51	24	19
Paper birch - black twinberry - common horsetail floodplain forest	undescribed	SH/bt	unlisted	Red	2	6	100	100	no data
Roche spruce - black cottonwood - hybrid alder - ostrich fern floodplain forest	ICHmc2/06	CD or SD	unlisted	Red	0				5
Black spruce - Roche spruce - scrub birch - sedge forested fen	ICHmc2/08	BB	unlisted	Red?	0				5

[a] Biogeoclimatic Ecosystem Classification (Banner et al. 1993)
[b] Terrestrial Ecosystem Mapping (Cadrin et al. 1996)
[c] Kispiox, Lakes and Morice Forest Districts, mapping very incomplete
[d] Percent by area in excellent or good condition
[e] Percent by area crown-owned land including protected areas, provincial forest, grazing tenures, and other unalienated land
[f] Bulkley District grasslands do not closely fit BW community description

representation of Saskatoon - slender wheatgrass ecosystems within newly approved or proposed Goal 1 and 2 protected areas, particularly in the Lakes District; however, active management of fire and herbivores, and some control of invasive species will be needed to maintain their diversity over time. We identified 2 variations of the ecosystem that are rare, highly threatened, and occur mostly on private land: (a) savanna-steppe with well-developed stands of Rocky Mountain juniper (*Juniperus scopulorum*; 15 known occurrences, with all but 2 on private land); (b) scrub-steppe ecosystems with deep soil on steep fluvial or lacustrine scarps. We recommended that the Saskatoon - slender wheatgrass community be down-listed to Blue status with special attention paid to occurrences on non-rocky soils, and that the Rocky Mountain juniper savanna-steppe receive separate status as a Red-listed plant community.

Bluegrass - Slender Wheatgrass Grasslands

True grasslands dominated by native bluegrasses (*Poa* spp.), slender wheatgrass (*Elymus trachycaulus*), and needlegrasses (*Stipa* spp.) are rare and highly endangered in the inland Skeena region. They are essentially absent from the Skeena watershed but exist on deep fluvial, lacustrine, and, occasionally, morainal parent materials along the north shorelines of major lakes of the Nechako Plateau. Much of the original grassland was submerged by the Nechako Reservoir in 1954, and most occurrences not under cultivation are now dominated by introduced Kentucky bluegrass (*Poa pratensis*),

timothy (*Phleum pratense*), and dandelion (*Taraxacum officinale*). Some of the best remaining grasslands are found on steep slopes below basalt buttes where soils are deep and rich, but too steep for intensive livestock use, and too dry and exposed for tree invasion. These sites are characterized by abundant pasture sage (*Artemisia frigida*) and teem with animal life.

We mapped 5 grasslands that marginally fit the Bluegrass - slender wheatgrass description in the Bulkley District and located 14 occurrences in other forest districts. We strongly recommended that the Red-list designation be retained for all low-elevation grasslands, although further work is needed to properly describe and classify grassland and closely related herb meadow communities.

Other Endangered Grassland, Herb Meadow, and Scrub-Steppe Ecosystems

Within the Skeena watershed most low-elevation herbaceous plant communities are not true grasslands, but moist, rich meadows dominated by cow parsnip (*Heracleum sphondylium*). We proposed 2 new herbaceous communities for Red-listing: a dry to mesic montane meadow ecosystem with extremely variable species composition (tentatively named Aster - meadow rue - peavine - fireweed) occurring on southwest-facing slopes and on inactive fluvial terraces, and a bracken-dominated (*Pteridium aquilinum*) variation of the more common cow parsnip meadow found on ICHmc sites that appear to have a long history of anthropogenic burning (Table 1). A mid- to higher-elevation grassland type (proposed Blue List) dominated by timber oat grass (*Danthonia intermedia*) and spike trisetum (*Trisetum spicatum*) was identified on coarse-textured, frost-prone fluvial terraces.

Management

Management options for these non-forest ecosystems are complex and controversial. Eliminating livestock grazing will not recreate the presettlement plant community and may accelerate tree and shrub encroachment. Carefully developed range use plans that control the distribution and season of use by livestock to avoid overgrazing can potentially benefit wildlife, encourage native grassland species, prevent degradation of sensitive sites, and assist recovery of degraded areas. However, livestock grazing should probably not be used to reduce tree encroachment on sites that currently have few nonnative species. Prescribed burning should be reintroduced where possible, with careful monitoring to avoid increasing the dominance of nonnative species.

A system of large and small benchmark ecosystems was proposed for the southwest-facing shrub-steppe, grassland, herb meadows, and associated aspen communities to develop and monitor prescriptions for prescribed burning and controlled grazing. In intensely settled areas such as the Bulkley Valley, where most occurrences are on private land, a public awareness and stewardship program was recommended to encourage private landowners to conserve and restore these ecosystems.

FLOODPLAIN AND SWAMP FORESTS

Black Cottonwood - Red Osier Dogwood - Prickly Rose Floodplains

The Red-listed Black cottonwood - red osier dogwood - prickly rose ecosystem is dominant on the active floodplains of major rivers in the inland Skeena region. In the SBSdk and adjacent SBSmc subzones it is extensive along the Bulkley and Morice rivers but rare on smaller rivers and creeks and along the shores of major lakes. Most of the 227 mapped occurrences (1,363 ha) in the Bulkley District are on private or municipal land (78%) and in marginal to poor condition (64%; Table 1). Extant cottonwood stands on the Bulkley mainstem are threatened over the long term by flood control measures and bank erosion. High ungulate browsing levels and invasive nonnative plant species are also concerns. Fortunately, the Morice River occurrences (mostly unmapped) are in good to excellent condition and probably more than double the total area of this ecosystem. It is difficult to justify Red-listing the Black cottonwood - red osier dogwood - prickly rose plant community based on abundance alone, but the listing is probably warranted by the degree of threat to occurrences away from the Morice River and the critical value of the ecosystem as wildlife habitat.

Other Floodplain and Swamp Forest Ecosystems

There are a wide variety of other forested and non-forested riparian plant communities on low-elevation SBS floodplains. Most are undescribed, and all are less abundant than the Red-listed Black cottonwood - red osier dogwood - prickly rose community. We identified the Hybrid white spruce - common horsetail floodplain or swamp forest community (Pojar et al. 1984) as rare and highly threatened on floodplains and slow-moving stream systems in settled areas within the SBSdk subzone, although a broadly similar community is widespread across northern British Columbia. Only 36 occurrences (115 ha, median size 1.9 ha) of this distinctive ecosystem, rich in fire-sensitive, nonvascular, and invertebrate species, were located in the Bulkley District. Several other new riparian plant communities were tentatively described and proposed for Red- or Blue-listing.

In the mountainous ICHmc subzone, cottonwood floodplains are more common than in the SBS and less threatened by agriculture or settlement. However, some riparian and rich swamp forest communities unique to the coast-interior transition warrant special recognition. Unusual forms of hybridization, exemplified by the 3-way hybrid Roche spruce (*Picea sitchensis* x *glauca* x *engelmannii*) and hybrid red alder (*Alnus rubra* x *incana*), and large, well-developed

populations of peripheral or disjunct species such as western redcedar (*Thuja plicata*), skunk cabbage (*Lysichiton americanum*), ostrich fern (*Matteucia struthiopteris*), and agrimony (*Agrimonia striata*), are characteristic features. These outstanding ecosystems are naturally rare and are threatened because they invariably occur at major river intersections or on other unusual landforms that have long served as focal points for human activity.

Management

There are few occurrences of large river floodplains in proposed protected areas. The Morice River corridor is the best opportunity in the southeastern Skeena region to maintain the Black cottonwood - red osier dogwood - prickly rose and associated SBS plant communities in near-natural condition with unrestricted flooding regimes. Mapping and preparing strategies for the scattered occurrences of rare riparian and swamp forest ecosystems along lakeshores and other stream systems in the region must become a management priority. On the Bulkley River and its tributaries, opportunities for restoration of natural communities and hydrological processes are greatly restricted. Further alienation and disturbance of remaining Crown land occurrences must be prevented. Efforts to restore connectivity should begin by working with Fisheries Canada, regional and municipal governments, and private landowners to develop a greenway system to prevent further clearing, stabilize banks, restore riparian vegetation, and allow natural flooding of backchannels and lower benches wherever possible.

LISTING RARE AND ENDANGERED ECOSYSTEMS

The provincial Red and Blue lists of plant communities maintained by the CDC in British Columbia are a very important first step towards recognizing and protecting special ecosystems. However, the program is limited by its lack of legislative teeth. Of the 241 plant communities currently listed by the CDC (94 on the Red List), only 4, none in Skeena Region, are recognized under the Forest Practices Code Act of British Columbia (B.C. MOF and B.C. MELP 1999). Pending federal endangered species legislation makes no provision for endangered ecosystems. As it currently stands, the primary value of the listing program is the awareness it creates among land managers, politicians, and the public.

A major technical limitation, particularly for nonforested ecosystems, is the absence of a detailed, widely agreed upon classification framework. Our work in the inland Skeena region showed that most rare ecosystems were not listed because they have not yet been adequately described or classified, or because they represent an unrecognized subset of a more widespread and recognized entity, such as a biogeoclimatic site series. Existing classifications such as BEC (Biogeoclimatic Ecosystem Classification; Banner et al. 1993) provide broad treatment of the most common and economically important (mainly forested) ecosystems, while detailed classifications of nonforested ecosystems are geographically restricted (e.g., Brett et al. 1998).

A potential negative side-effect of the Red- and Blue-listing scheme is that it may focus management attention on a static and narrowly defined target (a "climax" plant association or old growth successional stage) when a more appropriate goal for long-term ecosystem management may be to maintain a dynamic ecosystem complex encompassing a spectrum of closely linked ecosystems at various successional stages.

COMMUNITY INVOLVEMENT

For the foreseeable future, we cannot expect strong legislation or an expanded protected areas network to save ecosystems at risk. In the inland Skeena region, as elsewhere, the rarest, most endangered ecosystems are chiefly located in settled, valley-bottom corridors, largely on private land, where few opportunities exist to restore a wilderness setting or create new protected areas. Humans, like other animals, find these warm, productive, and diverse environments irresistible.

It is no coincidence that all major municipalities in the study area (Smithers, Telkwa, Hazelton, Houston, and Burns Lake) are situated directly on, or are immediately surrounded by, 1 or more endangered ecosystems. In many ways each town derives its unique character and identity from that physical setting. The willingness of local residents to adopt endangered ecosystems as symbols of community and to take responsibility for their health and productivity will determine not only the future viability of these ecosystems, but also the beauty and vitality of their human settlements. As biologists and ecologists concerned about natural ecosystems and species at risk we have a crucial role to play in helping our local, human communities make this vital connection.

ACKNOWLEDGEMENTS

This 2-year project was funded by the Habitat Conservation Trust Fund of British Columbia and administered by the Skeena Region Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks.

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Maintaining Diverse and Dynamic Ecosystems on Southwest-Facing Slopes in the Skeena Region

Sybille Haeussler
Skeena Forestry Consultants
4279 Rue Boyer, Montreal, PQ, H2J 3C8, Canada
skeena@internet.uqam.ca

ABSTRACT

Southwest-facing slopes in the inland Skeena region support a diverse mosaic of ecosystems with many peripheral plant and animal species. Maintaining their very dynamic disturbance regimes while simultaneously avoiding environmental degradation presents a special challenge for ecosystem management.

Key words: disturbance dynamics, grassland, scrub-steppe, Skeena.

Warm, dry, southwest-facing slopes in the Sub-Boreal Spruce biogeoclimatic zone (SBSdk and SBSmc subzones) and Interior Cedar–Hemlock zone (ICHmc subzone; Banner et al. 1993) of the southeastern Skeena region of British Columbia present a special challenge for ecosystem management. These slopes support a rich mosaic of ecosystems with some of the highest biological diversity found in northwestern British Columbia, including many plant and animal species at the periphery of their natural ranges. Two Red-listed plant communities (*Juniperus scopulorum - Amelanchier alnifolia - Elymus trachycaulus* savanna steppe, and *Poa secunda - Elymus trachycaulus* grassland), several Blue-listed communities, and a variety of rare-but-undescribed plant communities are found within the complex of scrub, grassland, woodland, and rock outcrop ecosystems that characterize these slopes (Fig. 1; Pojar et al. 1984, Haeussler and Hetherington 2000). Other special values include high aesthetic, recreational, and real estate potential, agriculture and range capability, critical ungulate winter habitat, and many indigenous food plants.

The aspect of these ecosystems is between 180° and 270°, but most often faces southwest, and rarely extends to the southeast. The terrain unit is typically a knoll or ridge of bedrock overlain by shallow to deep colluvial, morainal, or fluvial deposits. A condensed topographic sequence can be found on steeply eroded fluvial or lacustrine escarpments ("river breaks") or canyon walls. Most occurrences in the Skeena region overlook a lake or river. At their best, southwest-facing slope complexes consist of a linked series of ridges and toeslopes, dissected by erosion channels, beginning in the subalpine Engelmann Spruce–Subalpine Fir biogeoclimatic zone and terminating with a fluvial or lacustrine escarpment at the valley floor.

Southwest-facing ecosystems experience the greatest range of temperature extremes of any ecosystem within the Skeena region (R. Trowbridge, B.C. Ministry of Forests, pers. comm.). They are the first to leaf out in spring and the first to change colour in fall. They accumulate the most heat in summer and are snow-free for much of the winter. They are often windy and have high evapotranspirative demand because of afternoon sunshine. Climatic extremes may be moderated by proximity to large bodies of water.

The exceptional biological diversity found on southwest-facing slopes results from a complex and highly active

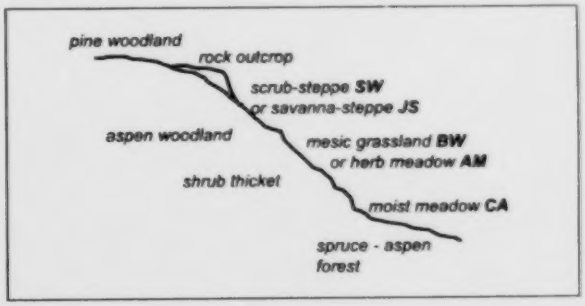

Figure 1. A topographic sequence of ecosystems on a southwest-facing slope in the Sub-Boreal Spruce SBSdk or SBSmc biogeoclimatic subzones. *SW* (*Amelanchier alnifolia - Elymus trachycaulus* scrub-steppe) and *BW* (*Poa secunda - Elymus trachycaulus* grassland) plant communities are currently Red-listed by the British Columbia Conservation Data Centre. *JS* (*Juniperus scopulorum - Amelanchier alnifolia - Elymus trachycaulus* savanna-steppe), *AM* (*Aster* spp. - *Lathyrus nevadensis - Thalictrum occidentale* mesic herb meadow), and *CA* (*Heracleum lanatum - Geum macrophyllum* moist herb meadow) are newly described, unlisted plant communities.

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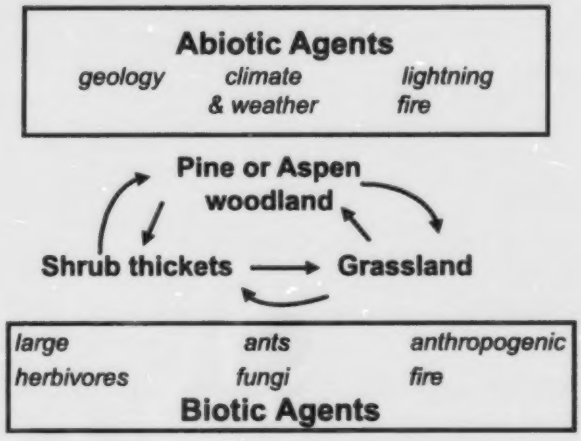

Figure 2. Schematic model of abiotic and biotic disturbance agents that maintain a dynamic mosaic of forest, shrub, and grassland vegetation on southwest-facing slopes.

disturbance regime operating at many scales of space and time (Fig. 2). Herbivores ranging in size from ants to large ungulates play a key role in preventing tree and shrub encroachment onto grassland. They also maintain grassland diversity through grazing pressure and by redistributing seeds and exposing mineral soil. Fires also profoundly influence ecosystem diversity (Fig. 3). On southwest-facing slopes, fires occur more frequently and are more variable in size, seasonality, and severity than in the surrounding landscape. Spring burning, in late March or April before leaf-flush, was used extensively by aboriginals and later by European settlers for land clearing and habitat manipulation (Poudrier 1892, Gottesfeld 1994), but accidental and lightning-caused fires in mid and late summer were also important. Today's mosaic of plant communities is the result of millennia of recurring, patchy fires of variable intensity. Short- and long-term climatic changes affect fire behaviour, as well as rates of tree encroachment and herbivore abundance.

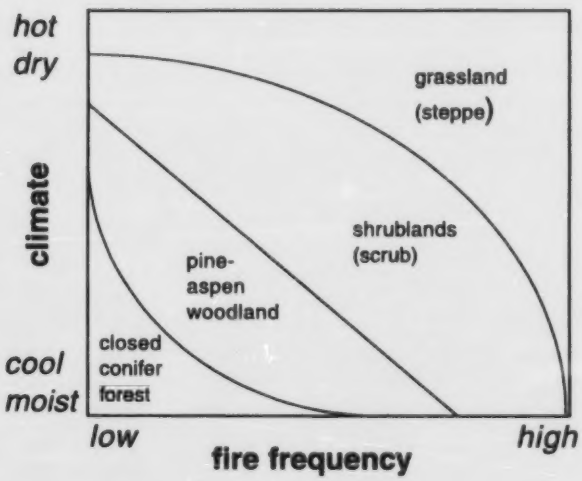

Figure 3. Effect of fire frequency and climate on southwest-facing slope vegetation. Frequent fires convert forest to shrubland, and shrubland to grassland. Fire suppression does the reverse. When or where the climate is hot and dry, grasslands are maintained at lower fire frequencies than when or where it is cool and moist.

Southwest-facing slopes are subject to many environmental threats including habitat fragmentation and loss from urban and industrial development, insensitive grazing practices, and recreational abuse. Paradoxically, it is their very dynamic nature that makes these ecosystems particularly vulnerable to invasive plant and animal species, and to changes to their disturbance regime such as fire suppression or overuse, mineral soil exposure, changes in herbivore abundance, and global warming. Management strategies to reduce ecosystem degradation must include building public awareness and interest, and must avoid a single-species or single-plant community focus. The Garry oak woodland and antelope-brush ecosystem conservation programs in southern British Columbia (Lea 1996, Spencer et al. 1997) face very similar challenges and can serve as models for preparing an effective ecosystem management strategy in the Skeena region.

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The Recovery of the Short-tailed Albatross and its Impacts on the North Pacific Longline Fishery

Tracee O. Geernaert
International Pacific Halibut Commission
P.O. Box 95009, Seattle, WA, 98145-2009, USA
tracee@iphc.washington.edu

Key words: longline fishery, Pacific Ocean, *Phoebastria albatrus*, short-tailed albatross.

EXPANDED ABSTRACT

The short-tailed albatross (*Phoebastria albatrus*), historically one of the most abundant albatross in the North Pacific, once numbered in the millions of birds and ranged over the entire Pacific Rim. Between 1887 and 1932, the feather trade brought the species to near-extinction. By 1950 they were considered one of the world's rarest birds, nesting on only a single volcanic Japanese island. In the 1960s, international protection allowed the population to begin recovery (Fig. 1). Though still considered endangered, today's population is approximately 1,000 birds. As the short-tailed albatross population recovers, the encounter rate in the North Pacific is increasing. In 1998, the Alaskan groundfish longline fishery killed 2 short-tailed albatross, half the allowable take for 1997–98. Since 1993 at least 7 short-tailed albatross have been reported killed by longline gear in Alaska.

Significant threats to the short-tailed albatross recovery include marine pollution (discarded plastic and chemicals), possible destruction of the nesting colonies from volcanic eruptions, and mortality from longline fishing (Fig. 2). Albatross are particularly vulnerable when longline gear is being set. Birds chase the baited hooks deployed from the stern of the vessels and may become hooked (Fig. 3).

In 1997, the North Pacific Longline Association recognized that short-tailed albatross bycatch posed a potential threat to their livelihood from possible legislation to protect the bird. The association led an effort to request regulations requiring seabird avoidance measures for the groundfish fleet in the northern Pacific Ocean off Alaska. Avoidance regulations covering the sablefish, Pacific cod, and rockfish longline fisheries

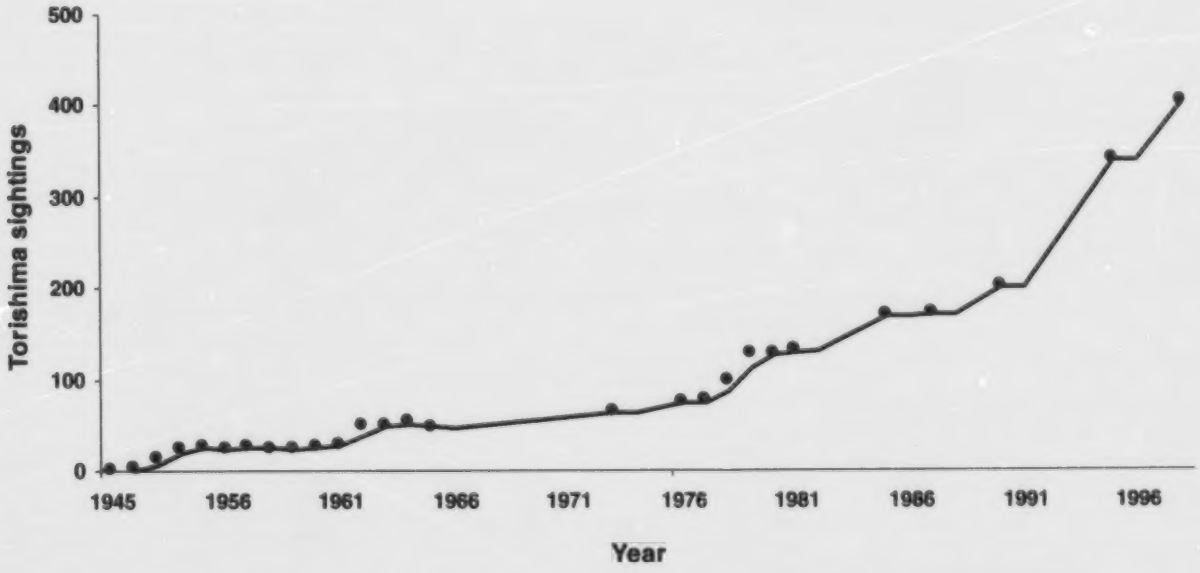

Figure 1. Recovery of the short-tailed albatross on Torishima Island, Japan. (Hasegawa, H. and A.R. Degrange. 1982).

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Table 1. Results from the 1998 International Pacific Halibut Commission halibut-vessel interviews on bird bycatch.

Data	British Columbia waters	Gulf of Alaska	Bering Sea and Aleutians	Total
Interviews conducted	941	2,689	481	4,111
Interviewed-pounds (millions)	10.4	29.4	6.2	
Interviewed-hooks (millions)	4.2	7.8	1.9	13.9
Birds caught/million hooks	7.1	8.6	0	6.1
Birds caught/million pounds	2.9	2.3	0	7.7
Short-tailed albatross observed	7	124	35	166
Blackfooted/Laysan albatross caught	21	11	0	32
Northern fulmars caught	3	54	0	57
Shearwaters caught	2	1	0	3
Others birds caught	4	1	0	5

were implemented in 1997. In 1998, similar regulations were implemented for the Pacific halibut fishery. Evaluation of the effectiveness of these measures is underway. New regulations are being drafted to extend the seabird avoidance measures to other fisheries. As part of the 1997–98 regulations a quota was set on the allowed "take" in both the longline groundfish and halibut fisheries. The limit set for each fishery applies over a 2-year period. A 2-bird limit was set in the halibut fishery, and a 4-bird limit was set for the other longline groundfish fisheries. If these limits are reached, the federal agencies involved could implement time-area closures or shut down fisheries altogether. The United States Fish and Wildlife Service (USFWS) is petitioning the United States government for domestic listing of the short-tailed albatross, which will afford it powerful protection.

In 1998, at the request of the USFWS and the North Pacific Fisheries Management Council (NPFMC), the International Pacific Halibut Commission (IPHC) agreed to interview the halibut fleet about seabird bycatch and avoidance measures. This provided the first seabird-bycatch data available for the North Pacific halibut fleet. More than 4,111 logbook interviews were conducted in both Alaska and British Columbia, representing 46 million pounds of halibut landed and almost 14 million hooks (Table 1). The highest reported bird bycatch occurred in the Gulf of Alaska, with 8.6 birds per million hooks from 29.4 million interviewed-pounds of halibut landed. British Columbia fishermen reported 7.1 birds caught per million hooks from 10.4 million interviewed-pounds landed. Seabird avoidance regulations

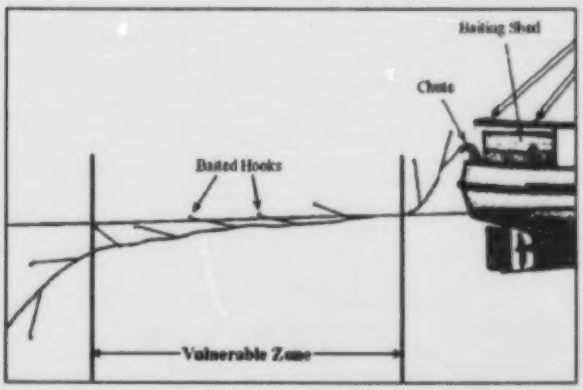

Figure 2. The potential hooking zone or vulnerable zone of albatross while fishing gear is being set.

Figure 3. An albatross hooked on longline gear from Alexander et al. (1977).

are not in effect in Canadian waters. The bird most frequently hooked in Alaska was the northern fulmar (54 birds), while only 3 were taken in British Columbia. The bird most commonly hooked in British Columbia was the albatross (either black-footed or Laysan), with 21 individual birds reported taken on longline gear. This was double the reported catch in Alaskan waters. Shearwaters and alcids were not caught in any significant numbers on halibut gear. No short-tailed albatross mortality was reported during the interviews, though >160 birds were observed by the fleet.

Tori lines and buoy bags were the most common and effective avoidance devices used in Alaska. Tori lines have streamers of strapping tape or rubber tubing spaced between the buoy bag and the vessel. The streamers act as a secondary scaring device along with the buoy bag. A towed device is most effective when placed aft of the bait's point of entry into the water to protect the zone of sinking hooks. If the device keeps the seabirds from attacking the bait they cannot become hooked. Other avoidance devices listed included towed boards, groundline weights, and noise makers. Many vessels used a combination of devices or alternated measures. Avoidance devices have been shown to increase catch of target species by allowing more bait to reach the ocean bottom (Bjordal and Lokkeborg 1996). The IPHC vessel interviews will continue in 1999.

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Sensitive Ecosystems Inventory Project for East Vancouver Island and the Gulf Islands

Jan Kirkby

Conservation Data Centre, Research Inventory Branch, Ministry of Environment, Lands and Parks
P.O. Box 9344, STN PROV GOVT, Victoria, BC, V8W 9M1, Canada
Jan.Kirkby@gems9.gov.bc.ca

ABSTRACT

The Sensitive Ecosystems Inventory (SEI) of East Vancouver Island and the Gulf Islands was initiated in 1993 in response to an urgent need for inventory information on remaining natural areas to support sound land-use planning. The eastern coastal lowland of Vancouver Island and the adjacent Gulf Islands were chosen as a pilot study area for the project because of their exceptionally high biodiversity values. This area contains some of British Columbia's rarest and most endangered plants, animals, and plant communities. It is also a highly desirable place to live, and is therefore experiencing intense development pressure. The objective of the SEI project was to provide mapped ecosystem inventory data to local governments, developers, landowners, and others as a user-friendly flagging tool for identifying remaining natural areas and important wildlife habitats, and to engage and encourage their participation in the protection of sensitive ecosystems in their area. The SEI project systematically identified, mapped, and evaluated the ecologically significant and relatively unmodified sensitive terrestrial ecosystems remaining in this area, specifically, wetland and riparian ecosystems, older forests and woodlands, coastal bluffs, sparsely vegetated dunes, spits and cliffs, and terrestrial herbaceous ecosystems. Seasonally flooded agricultural fields and large, older, second-growth forests were also mapped for their general biodiversity values. The inventory results strongly support claims that natural areas and habitats are fast disappearing. Mapped inventory data show that <8% of the entire study area can be considered relatively natural, and many of these areas have been substantially degraded by fragmentation, human use, and introduced species. SEI project staff are now working with local governments, landowners, developers, and others to use existing tools and mechanisms to protect and conserve the sites identified.

Key words: Gulf Islands, inventory, land-use planning, Vancouver Island.

The eastern coastal lowland of Vancouver Island and the adjacent Gulf Islands comprise a unique ecological region in Canada, with exceptionally high biodiversity values and many rare and endangered plants, animals, and plant communities. Intense development pressure in this area has resulted in the fragmentation and loss of these rare ecosystems. The Sensitive Ecosystems Inventory (SEI) of East Vancouver Island and the Gulf Islands was initiated in 1993 to provide an inventory information base on remaining natural areas that would support sound land-management decisions and promote good land stewardship.

The Sensitive Ecosystems Inventory is a joint project of Environment Canada (Canadian Wildlife Service) and the British Columbia Ministry of Environment, Lands and Parks (Conservation Data Centre and Vancouver Island Region), with funding support from the Habitat Conservation Trust Fund and several local governments. The purpose of the SEI project was to systematically identify, map, and evaluate the ecologically significant and relatively unmodified sensitive terrestrial ecosystems remaining in the study area, specifically:

- coastal bluff – vegetated rocky islets, shorelines, and coastal cliffs;
- sparsely vegetated – sand dunes, spits, and inland cliffs;
- terrestrial herbaceous – mosaics of coastal grassland meadows and moss-covered outcrops;
- wetland – marshes, fens, bogs, swamps, shallow water, and wet meadows;
- riparian – vegetated floodplains, lakeshores, and gullies;
- woodland – open forests dominated by deciduous trees with canopy cover generally <50%;
- older forest – forested units older than 100 years.

These 7 broad ecosystem types represent the remaining natural ecosystems within the study area that are considered to be particularly rare, fragmented, or threatened by

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continued development. These ecosystem types also contain higher concentrations of rare plants, animals, and plant communities than are found in other ecosystem types within the study area. Seasonally flooded agricultural fields and large, older, second-growth forests were also mapped as part of this inventory for their general biodiversity and wildlife habitat values.

The study area was based on the Nanaimo Lowland ecosection boundary; it extends from Campbell River in the north to Sooke in the south and includes the islands in the Islands Trust (Southern Gulf Islands and Strait Of Georgia ecosections), with a total area of roughly 4,000 km^2 (Fig. 1). Sensitive ecosystems were first identified on aerial photographs at scales ranging from 1:10,000 to 1:20,000. A representative sampling (approximately 30%) of the identified sites was then field-checked to verify boundaries, and to classify, photograph, and evaluate present conditions. Information on landscape condition, polygon uniformity, environmental characteristics, site association, disturbance history, adjacent land use, known threats, and wildlife observations was gathered, as well as a list of dominant plant species and percent cover. Cover values of introduced plant species were also noted to assist in determining site quality and condition.

The air photos were digitized and >7,000 sites were mapped on a TRIM (Terrain Resource Information Management) base. Sixty-six maps at 1:20,000 were developed using ArcInfo GIS (geographic information system) and are available in both digital and hardcopy formats. Digital coverage on compact disc was made available to those agencies with GIS capability. Maps were supplied at no cost to all local governments within the study area. The maps and accompanying database provide a valuable flagging tool for identifying remaining natural areas and important wildlife habitats.

The inventory results (Fig. 2) strongly support claims that natural areas and habitats are fast disappearing. Today, <8%

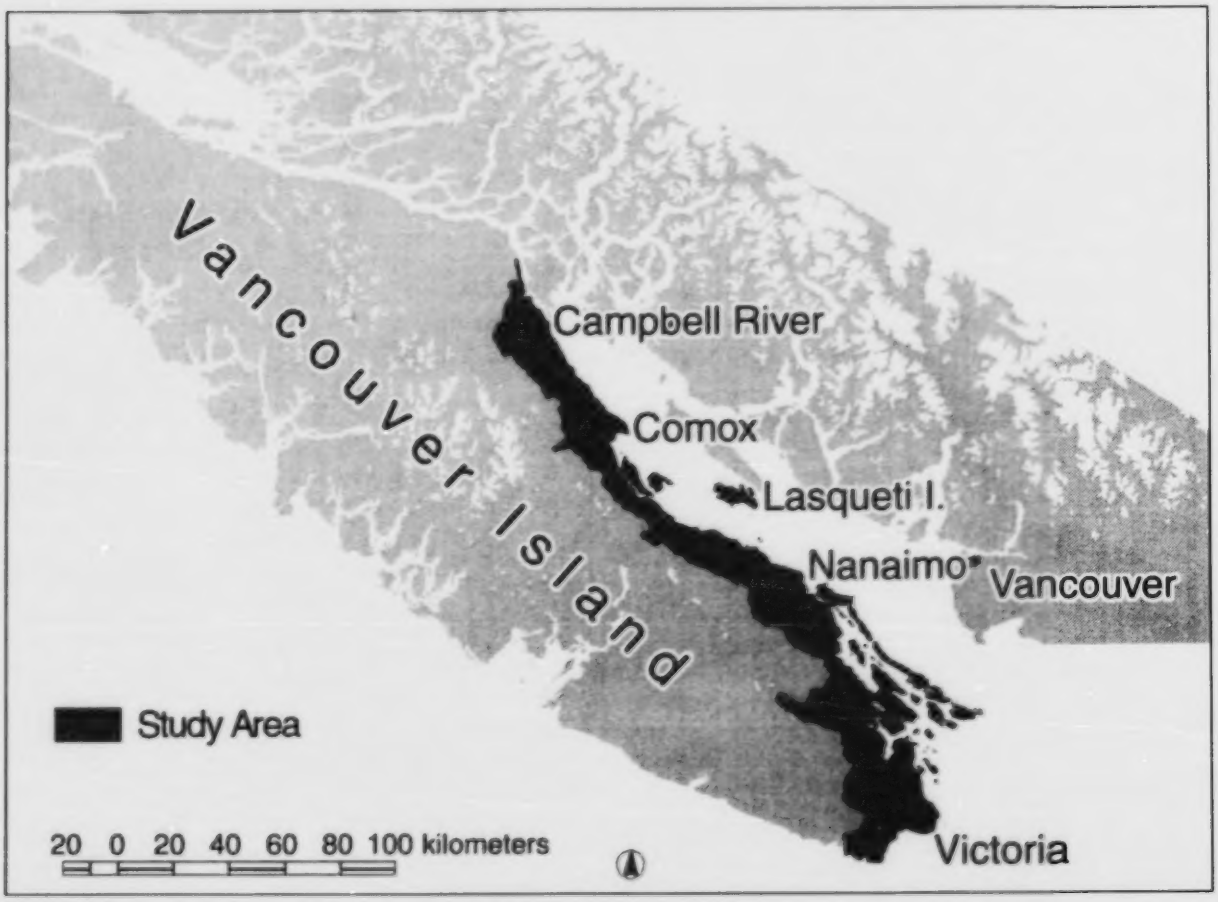

Figure 1. Sensitive Ecosystems Inventory (SEI) study area.

of the entire study area can be considered relatively natural, and many of these areas are substantially degraded by fragmentation, human use, and introduced species. An additional 0.7% of the study area was mapped as seasonally flooded agricultural fields, with another 10.9% occupied by large stands of second-growth forests >60 years old. The rest of the study area is under urban or rural land use, or comprises recent clearcuts or young forests of <60 years of age.

The second stage of the SEI project focused on the development of support materials and extension services. Orientation and information sessions were offered to provide information on how SEI data can be used in land-use planning in order to ensure the continued integrity of these fragile and threatened ecosystems.

SEI project staff are now working with local governments, landowners, developers, and others to use existing tools and mechanisms to protect and conserve the sites identified. Two publications have been developed:

- a technical report describing the project and summarizing and analyzing the data; and
- a conservation manual intended for use by local governments, developers, landowners, and conservation agencies to engage and encourage their participation in the protection of sensitive ecosystems in their area.

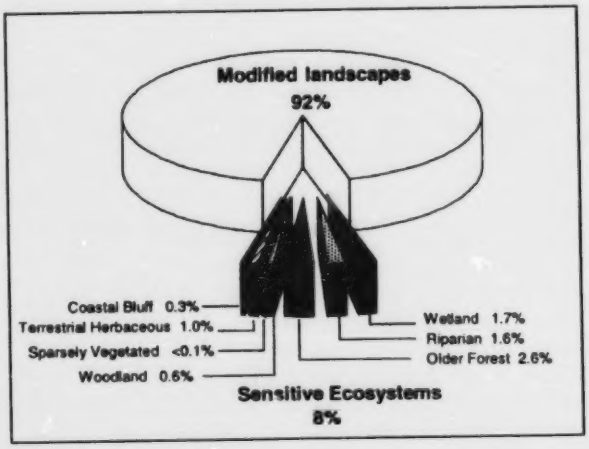

Figure 2. Inventory results.

The response to the SEI data has been gratifying. The large number of requests for SEI data demonstrates the need for the specific ecological data generated by this study. The SEI Technical Advisory Group has offered user support to a wide variety of groups and individuals, answering an average of 15 data requests each month from land trusts, conservancy groups, regional and municipal governments, contractors, developers, First Nations, educational institutions, and others. SEI data have already been used by several local governments in developing and/or revising their official community plans, and the Capital Regional District has used SEI data in the development of their Green and Blue Spaces strategy. Interpretation of SEI data from a regional and local perspective has also been offered by SEI project staff, providing scientific support to planners and local governments in the designation of Development Permit Areas (DPA) and the development of conservation strategies for their areas.

SEI data has helped to bring attention to sites of conservation priority, and has also been used in ecoregional planning projects. Conservation evaluations based on SEI field data and Conservation Data Centre element occurrence data have provided solid scientific support for establishing conservation priorities for land acquisition and conservation. Information and support on SEI methodology and procedure have been offered to groups embarking on similar inventory projects in the province.

The ecological information provided by the Sensitive Ecosystems Inventory provides the basis for a comprehensive conservation strategy for East Vancouver Island and the Gulf Islands. The successful model of the Sensitive Ecosystems Inventory will provide the framework for similar inventory projects in other parts of British Columbia similarly threatened by intense development pressure. SEI projects are currently planned for the Sunshine Coast and the central Okanagan in an effort to provide scientific input to encourage sound conservation-based land-use planning decisions.

Inventory of the Queen Charlotte Islands Ermine

Donald G. Reid
Fish and Wildlife Branch, Ministry of Environment, Lands and Parks
P.O. Box 5000, Smithers, BC, V0J 2N0, Canada
Don.Reid@gems6.gov.bc.ca

Louise Waterhouse
Research Section, Ministry of Forests
2100 Labieux Road, Nanaimo, BC, V9T 6E9, Canada

Peter E. F. Buck
Systems Forestry Consulting,
P.O. Box 442, Masset, BC, V0T 1M0, Canada

Andrew E. Derocher
Norwegian Polar Institute
Storgt. 25, Boks 399, 9001 Tromso, Norway

Rolf Bettner
P.O. Box 495
Queen Charlotte City, BC, V0T 1S0, Canada

Colin D. French
P.O. Box 812
Queen Charlotte City, BC, V0T 1S0, Canada

ABSTRACT

A subspecies of ermine (*Mustela erminea haidarum*) is one of the few endemic terrestrial mammals on the Queen Charlotte Islands (QCI). This subspecies is on the British Columbia Red List, and is classified as Vulnerable by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). Its status reflects its rareness on QCI historically, the paucity of mammalian prey species available, and lack of knowledge of its habitat use and population limitations. Competition from an increasing endemic marten (*Martes americana*) population may be threatening ermine population viability. From 1992 to 1998 we used a variety of inventory techniques, over a range of habitats, to determine QCI ermine distribution and habitat use as follows: live-trapping with Sherman traps (1992 summer; 1993 fall; 1995 spring; 1997 spring); interviews with QCI trappers and residents (1997 and 1998); snow-tracking and track stations (1997 and 1998). Between 275 and 325 interviews produced 162 new records (mainly sightings) to add to 32 previous records. Live-trapping was less successful (1992: 0 in 50 trap nights; 1993: 2 in 2,301 trap nights; 1995: 0 in approx 3,000 trap nights; 1997: 0 in 1,414 trap nights). Snow-tracking (22 km on foot; 900 km by road) and track stations (2,692 nights) produced no definite ermine sign. The great majority of new records were from the Coastal Western Hemlock submontane wet hypermaritime biogeoclimatic variant (CWHwh1), with ermine most often found within 100 m of a water body, including the ocean. Ermine were live-trapped in second-growth forest. Results demonstrate a widespread distribution on the archipelago (including some smaller islands), and a continuing rarity. They suggest no particular reliance on old-growth forest, but heavier use of riparian and marine foreshore habitats.

Key words: ermine, interspecific competition, introduced species, inventory, marten, *Martes americana*, *Mustela erminea*, Queen Charlotte Islands, short-tailed weasel, stoat.

The Queen Charlottes Island ermine or short-tailed weasel (*Mustela erminea haidarum*) is limited in distribution to the Queen Charlotte Island (QCI) archipelago off the central British Columbia coast. This subspecies exhibits less sexual dimorphism than other subspecies, and unique cranial morphology (Foster 1965). It appears to have persisted, separate from mainland ermine populations, in a coastal refugium isolated by ice during at least the most recent Pleistocene glaciations, and subsequently to have been isolated by the sea (Foster 1965, Cowan 1989).

Prior to European settlement, only 8 species of non-marine mammals persisted on QCI, and this fauna was dominated by carnivores including the ermine, a subspecies of pine marten (*Martes americana nesophila*), and a subspecies of black bear (*Ursus americanus carlottae*; Foster 1965, Nagorsen 1990). The dominance of carnivores likely resulted in intense competition for the few endemic mammalian prey: dusky shrew (*Sorex monticolus elassodon*); deer mouse (*Peromyscus maniculatus keenii*); and Sitka mouse (*Peromyscus maniculatus prevostensis*; Nagorsen 1990). This may explain the relative rarity of the carnivores and their tendency to forage in the intertidal (Foster 1965). Since the early 1900s, the introduction of black-tailed deer (*Odocoileus hemionus columbianus*), red squirrel (*Tamiasciurus hudsonicus lanuginosus*), raccoon (*Procyon lotor vancouverensis*), muskrat (*Ondatra zibethicus*

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

osoyoosensis), roof rat (*Rattus rattus*), domestic cat (*Felis domesticus*), and beaver (*Castor canadensis leucodontus*; Nagorsen 1990) has altered the patterns of energy flow through the food web in ways that are not altogether clear (Cowan 1989). The non-marine mammalian fauna on QCI remains depauperate (11 species) compared to the mainland coast at the same latitude (31species), and is still dominated by carnivores (Cowan 1989). It is conceivable that some carnivores, such as the marten and raccoon, have benefited from the more diverse, introduced prey base. However, the smaller ermine, often not adapted to feed on these introduced species, may face more intense competition from proliferating populations of competitors.

The QCI ermine is a Red-listed ("endangered/threatened") species in British Columbia. It is classified as Vulnerable nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). These rankings reflect its apparent rarity, the lack of knowledge of its habitat use and population limitations, and its likely vulnerability. This vulnerability may have increased because of major changes in the pattern of habitat distribution on QCI following European settlement activities such as timber harvesting, trapping, and agriculture, and because of changes in the species composition and energy flow through the food web following mammalian introductions.

Given this general knowledge, we initiated an inventory in 1992 with the goal of better assessing the current status of the QCI ermine. Our specific objectives have been to: 1) more accurately determine its present distribution on QCI; 2) obtain some estimate of abundance and historical trend in abundance; and 3) determine its habitat associations. The inventory work has been coordinated by A. Derocher, D. Reid, and L. Waterhouse over the years. Much of the field work has been accomplished by P. Buck, R. Bettner, and C. French, working as contracted biologists, and other staff as acknowledged below. This paper summarizes the inventory work to date, and suggests some routes for further investigation.

STUDY AREA

The Queen Charlotte Island archipelago is separated from the mainland coast by a minimum of 80 km (Cowan 1989), and has 2 main islands, Graham and Moresby, separated by narrows. Biological and geological evidence indicates that unglaciated refugia, largely tundra, persisted on the present land mass of QCI and on exposed portions of the continental shelf through the late Wisconsin glaciations (Heusser 1989). Whether these refugia were sufficient in extent and biotic composition to enable the persistence of ermine populations, isolated from mainland populations throughout the Pleistocene, remains unclear. However, isolation would have been absolute since the retreat of the latest Wisconsin glaciation, and subsequent inundation of the continental shelf in Hecate Strait (Foster 1965, Heusser 1989).

Today, the QCI archipelago has a maritime climate. Winter snowfalls are relatively uncommon, and the snow rarely persists below 300 m elevation for more than a few days. Terrestrial habitats are primarily forested. The Coastal Western Hemlock (CWH) zone ranges from sea level to 600–700 m. The Mountain Hemlock (MH) zone extends above the CWH up to 800 m (lower on windward slopes and higher on leeward slopes), and Alpine Tundra (AT) covers higher elevations than the MH (Meidinger and Pojar 1991). These zones are subdivided into CWHwh1 (submontane wet hypermaritime), CWHwh2 (montane wet hypermaritime), CWHvh2 (central very wet hypermaritime), MHwh1 (windward wet hypermaritime), and MHwh2 (leeward wet hypermaritime; Green and Klinka 1994). Extensive land clearing for agriculture and timber harvesting has dramatically increased the proportion of younger seral stage vegetation communities on the islands, especially in the past 50 years. In addition, there are numerous beach, bog, marsh, and fen habitats on the archipelago.

METHODS

Information on distribution, abundance, and habitat use was gathered, often at the same time, using the following techniques: review of previous records; interviews regarding new records; live trapping; track station surveys; snow tracking.

Previous Records

We compiled records existing previous to this study. These included records in the Conservation Data Centre of the British Columbia Ministry of Environment, Lands and Parks and records in the Royal British Columbia Museum. The latter included this museum's specimens, and records of specimens held in the American Museum of Natural History, the Cowan Vertebrate Museum at the University of British Columbia, the Smithsonian Institution, and the Museum of Vertebrate Zoology at the University of California, Berkeley and Los Angeles.

Interviews

We conducted detailed interviews with holders of registered commercial traplines on QCI, and interviews with other QCI residents and visitors known to have seen, or have been likely to see, ermine or ermine sign. Key interviewees were identified by long-time trappers and QCI residents George and Terry Husband. Many of the interviews were solicited through newspaper articles and advertisements in July and August 1997 respectively, and an information booth and display at the Tlell Fall Fair in August 1997.

All interviews were conducted using a formal Occurrence Report Interview form, recording the following information: 1) observer name and contact; 2) date of observation; 3) location

with UTM (universal transverse mercator) coordinates where possible; 4) elevation; 5) type of observation (visual, trapped live, trapped dead, snow track, other track, other sign); 6) broad habitat type (coniferous forest, deciduous forest, mixed forest, shrub, fen, bog, pasture, road, clearcut, beach); 7) forest harvest history (clearcut <10 yr, clearcut >10 yr, selective cut <10 yr, uncut or selectively cut >10 yr, unknown); 8) nearest water type (marine, estuary, lake, river >3 m wide, creek <3 m wide, marsh, unknown); 9) distance to nearest open water; 10) observation details confirming an ermine (size, pelage, tracks, etc.). We asked observers to describe the animal in detail, and we were looking for a description including at least 1 of the following characteristics: elongate body with long tail, short legs, and small head; rusty brown above with white belly in summer, and white in winter; black tip on the tail; bounding, smooth, and slinking gait. We asked trappers a number of more detailed questions regarding their history of trapping, their choice of bait, and their impression of the status of ermine.

Live Trapping

We attempted to capture ermine alive in July 1992, September 1993, March and April 1995, and February and March 1997. We used double-door Tomahawk cage traps (#206; Tomahawk Live Trap Co., Tomahawk, WI) in 1992, and folding aluminum Sherman traps (model XLF15; H.B. Sherman Inc., Tallahassee, FL) with dimensions 10 X 12 X 38 cm, in subsequent years. The following baits and lures were used: 1992, raw deer and bacon; 1993, raw bacon and Hawbaker weasel lure (S. Hawbaker and Sons, Fort Loudon, PA); 1995, raw bacon, and beaver meat and castors; 1997, sardines and fish fertilizer.

In 1992, traps were set primarily in riparian habitats, spaced every 500–1,000 m, and checked once a day for periods of 4 days. In subsequent years, traps were set in lines 500–2,000 m long, at sites chosen on the basis of past sightings, and with a view to sampling a range of habitat types. In these years, traps were approximately 50 m apart, and were placed at the base of trees, and under logs and tree roots. They were checked twice daily for a period of 4 days. Because of high capture rates of non-target species, principally the 2 species of *Peromyscus* (hereafter called deer mice), the 1997 effort included placement of either 2 or 4 traps at each station. Habitats were defined as follows: mature forest (>80 yr); second growth (10–80 yr); clearcut (<10 yr); marine shorefront (any within 100 m of high tide); riparian (any within 50 m of freshwater); bog (treed with moss and standing water); marsh/fen (standing water with emergent vegetation, often graminoids, but no canopy).

In 1992, traplines were located on Moresby Island near Alliford Bay and on Graham Island at Survey Creek. In 1993, traplines were located in numerous locations on Graham Island including Yakoun Lake, Rennell Sound, Honna River, Delkatla marsh, Tow Hill Ecological Reserve, White Creek, Pure Lake, Mamin River, lower Yakoun River, Tlell River; and at numerous sites on Moresby including Sachs Creek, Alliford Bay, Gray Bay, Copper Creek, Skidegate Lake, and South Main road at miles 1 and 4. In 1995, traplines on Graham Island were located at Kumdis Creek, Yakoun River, Gold Creek, Woodpile Creek, Lawnhill, and Honna River; and on Moresby Island at Sachs Creek, Moresby Camp, and Mosquito Lake. In 1997, traplines on Graham Island were set at Tarundl Creek and Kagan Bay, and on Moresby Island at Sachs Creek and South Bay Main road.

Track Stations

We attempted to document ermine distribution and habitat use by attracting ermine to baited and scented track stations from October 1997 through February 1998. Each station consisted of a Sherman aluminum live trap, previously boiled in a solution of boughs of local trees, and placed, generally under substantial cover of logs, tree roots, or boughs, as a cubby with 1 door locked open. The tracking plate was an aluminum sheet cut to fit on the floor of the trap cubby, with chevron cuts to hold white tracking paper close to the mouth of the cubby, and carbon soot placed on the plate in the interior of the cubby, following procedures outlined by Zielinski (1995). Each cubby was baited with a spreadable mixture of fish oil, butter, and Hawbaker weasel lure (S. S. Hawbaker and Sons, Fort Loudon, PA), and also, in 1998, commercial ermine anal scent gland semiochemical (1:1 mixture of 2-proplythietane and 3-propyl-1,2-dithiolane; Phero Tech Inc., Delta, BC). In addition we sprayed a solution of fish oils and weasel lure in alcohol on local vegetation in attempts to disperse the odours more widely.

Track stations, about 50 m apart, were set in lines traversing a variety of habitats, with special emphasis on riparian and marine foreshore habitats in areas where ermine had previously been reported. To change the sooted carbon plate and tracking paper, stations were initially visited every day and were kept in place for only 4 days. In November 1997 this period was increased to 7 days, with visits every second day.

Snow Tracking

We attempted to document ermine distribution and habitat use in winter 1998–99 by travelling on foot or by vehicle through forested habitats or along roads approximately 2 days after a snowfall.

RESULTS

Distribution

Thirty records of QCI ermine previous to this study indicated an extensive distribution on Graham Island, and a less extensive distribution on Moresby Island, but no confirmed evidence from other smaller islands (Appendix A). Only 1

record, from Skidegate Channel, indicated a distribution including portions of the west coast of the archipelago. Ninety percent of previous records were of ermine trapped dead by registered trappers, or other collectors.

We collected 162 new records of QCI ermine from interviews (Fig. 1, Appendix B). Each record represents an individual ermine, or evidence of 1 ermine. Of these, 121 (75%) were visual observations. Nearly all observers were able to identify at least 1 of the key characteristics of an ermine, and 69 of the visual observations (57%) were of ermine with the distinctive white pelage of winter. Twenty-five records (15%) were of ermine trapped dead by registered trappers, and not previously reported to the Conservation Data Centre. Seven (4%) were of snow tracks, with 5 reported by experienced QCI trappers. Six reports (4%) were based on other sign: 3 of dead chickens killed by ermine based on skin punctures; 2 of secondhand reports of cabin pets; 1 of an ermine den identified by odour. Three (2%) were of ermine killed by domestic cats. We consider these records complete to the end of 1997.

New records indicated a distribution encompassing 2 additional islands of substantial size in the archipelago: Louise (records from 1983, 1986, and 1993); Burnaby (record from 1985; Fig. 1, Appendix B). One record was of an ermine on a floating log boom (Sep 1980, Skaat Harbour, Moresby Island; Appendix B). Occupation of relatively isolated habitats on the west coast was confirmed for Tian Head, Rennell Sound, and Government Creek.

New records were predominantly from portions of the archipelago with the highest human populations (near the settlements of Masset, Port Clements, Tlell, Skidegate, Queen Charlotte City, Alliford Bay, and Moresby Camp), or from along the roadways linking these settlements. However, distribution was not even along these roadways. Records were clumped in the following locations on Graham Island: Delkatla and Canadian Forces Base Masset; middle Masset Sound near mouth of Watun River; Kumdis Creek; tributaries to southern Mayer Lake; Mayer River; lower Yakoun River; confluence of Gold Creek and Yakoun River; lower Phantom Creek and Yakoun River; Lawn Creek and Lawnhill coast; Tarundl Creek and lower Honna River. On Moresby Island these include: lower Sachs Creek and Alliford Bay; the stream draining into Skidegate Inlet along the South Main road; Mosquito Lake and Creek; Copper Creek and Copper Bay. These records indicate an association with water bodies, both freshwater and marine.

Intensive live trapping produced few results (Table 1). We caught only 2 ermine. Both were caught in mid-September 1993 on Moresby Island, 1 on lower Sachs Creek, and the other near mile 1 on the South Main road. Both of these were sites identified by interviews as likely locations for ermine. We caught numerous deer mice in the Sherman live traps (Table 1).

Track stations, run for a total of 2,692 station nights, produced no confirmed evidence of ermine. Track station lines were run in the following areas: lower Tarundl Creek and Honna River (Oct: 196 station nights); Sleeping Beauty (Nov: 171 station nights); Sandspit to Grey Bay (Nov: 215 station nights); Rennell Sound (Nov: 277 station nights); Delkatla to Rose Spit (Nov: 347 station nights); Tlell River to Mayer Lake (Jan: 370 station nights); Lawnhill (Feb: 140 station nights); Yakoun River (Feb: 472 station nights); Deena River (Feb: 240 station nights); Government Creek (Feb: 264 station nights).

Searches for snow tracks covered approximately 22 km on foot and 900 km by vehicle on roads, but produced no evidence of ermine.

Abundance

Our interviews revealed records of ermine from 1920 to 1997. Where age could be attributed accurately to decade (n = 161), there was a moderate frequency in the first 3 decades (1920s: 15; 1930s: 9; 1940s: 11), few in the 1950s (1) and 1960s (6), and an increasing frequency in recent decades (1970s: 24; 1980s: 37; 1990s: 58). Over the past decade there were 1–13 records per year. The decline in frequency of observations in the 1950s and 1960s is intriguing. The proportion of records of ermine trapped dead prior to 1950 (n = 12, 34%) was substantially higher than the proportion since 1950 (n = 13, 10%). Conversely, the proportion of visual observations increased from 51% (n = 18) before 1950 to 81% (n = 102) after 1950.

On 5 occasions, 2 ermine were seen together. In 1 case, both were believed to be juveniles, judging by their small size.

Habitat Associations

New Records

To investigate the ermine's habitat associations based on data from interviews, we removed records in the following circumstances: the ermine had been trapped without clear recollection by the trapper of habitat (n = 7); a second ermine was seen at the same time, so only 1 of the 2 records was used (n = 5); a cat killed the ermine (n = 3); the ermine was in a building including a chicken house (n = 17; Appendix B). This left 130 new records for the analyses.

Nearly all records (n = 121, 93%) were from the CWHwh1, the biogeoclimatic variant covering most of the eastern side of the archipelago below approximately 350 m. There were 6 records from the CWHvh2, the very wet variant at lower elevations on the west side of the Windward Mountains; 2 records from the CWHwh2, the variant above 350 m elevation on the east side of the Windward Islands; and 1 record from the MHwh2, near tree line. There were no records from the MHwh1 or from the Alpine Tundra.

The great majority of the 130 records (87%) were associated with forested habitats, including coniferous (69%), mixed (9%), deciduous (1%), shrub (1%), and bog (7%) types.

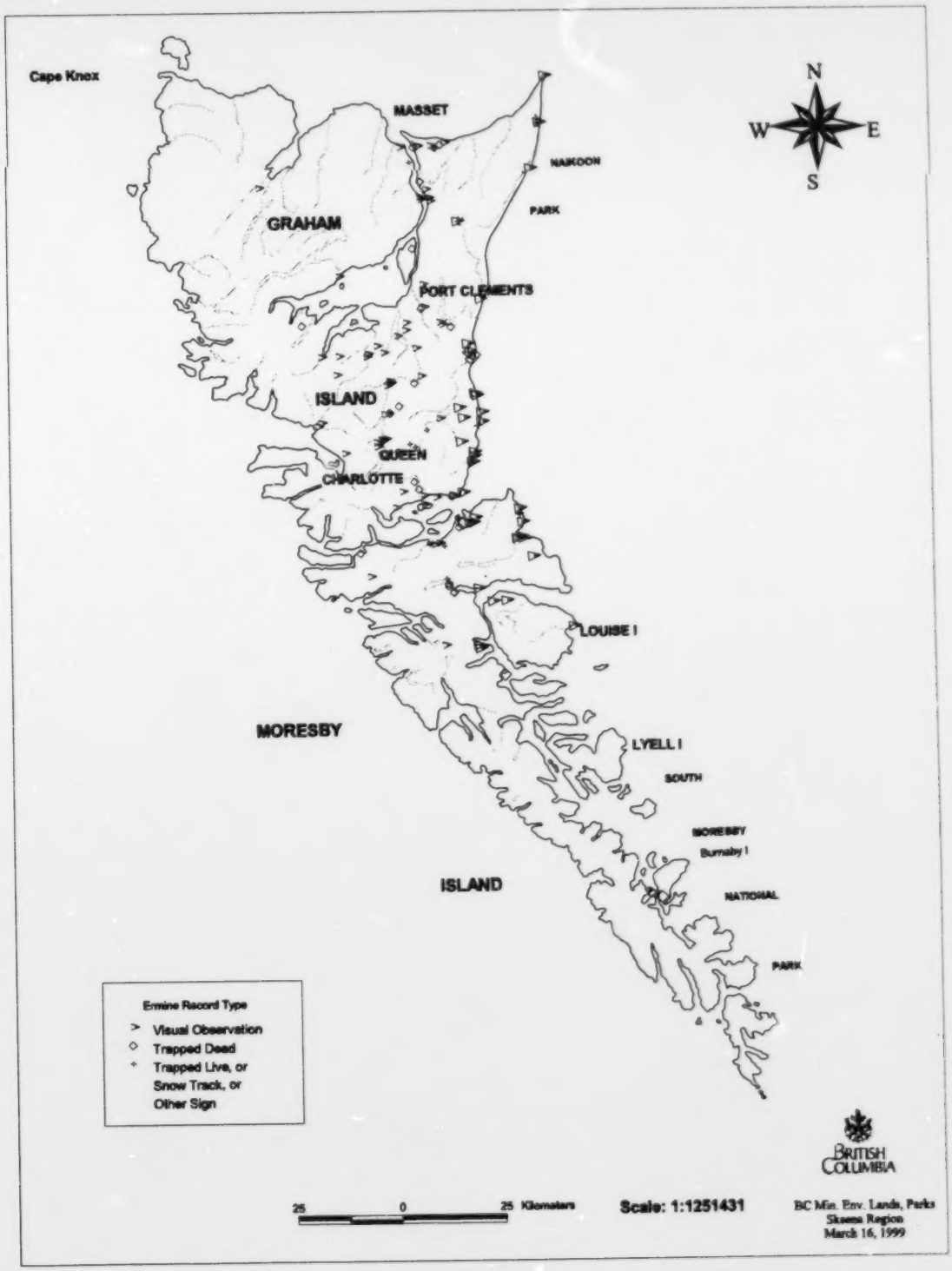

Figure 1. Locations of new records of Queen Charlotte Islands ermine by record type.

Table 1. Live-trapping dates, habitats, effort, and capture rates for Queen Charlotte Islands ermine and deer mice.

Date	Habitat type[a]	Trap-station nights[b]	Trap nights	Ermine captures	Deer mice/100 trap nights
1992 (8–28 Jul)	Riparian	100	100	0	not applicable
1993 (9–22 Sep)	Mature forest	220	220	0	13
	Second growth	445	445	1	9
	Clearcut	287	287	0	18
	Marine shorefront	456	456	0	23
	Riparian	552	552	1	9
	Bog	102	102	0	4
	Marsh/Fen	239	239	0	23
1995 (Mar)	Various	1,050	1,050	0	not available
1997 (24 Feb–6 Mar)	Riparian	490	1,414	0	25

[a] As defined in the text.
[b] Trap-station nights = total no. of stations X no. of nights.
[c] Trap nights = total no. of traps, including multiple traps per station, X no. of nights.

Five percent were associated with beaches, and pasture and fen habitats included 4% each.

Of the 113 observations associated with forested habitats, 54 (48%) were in uncut forest, 42 (37%) in clearcuts >10 years old, and 16 (14%) in clearcuts <10 years old.

One hundred and seventeen of the records included data on distance from nearest water body. Water body types nearest to an observed ermine included all the principal types on the archipelago, with ocean, creeks, and rivers dominating (Fig. 2). Ninety of 117 records (77%) were within 100 m of water (Fig. 2). There was a tendency for ermine to be closer to water when the water was the ocean, an estuary, a river, or a creek, than when it was a lake or marsh (Fig. 2).

Reasonably accurate estimates of elevation were available for 73 of the 130 observations. A substantial majority (n = 47, 64%) were within 10 m of sea level (high tide). These naturally included all sightings in beach habitats, as well as all sightings in pasture. They also included 33 of 56 (60%) records from forested habitats. A smaller proportion (n = 17, 23%) of records were from 11–50 m elevation, and fewer still (n = 9, 12%) were above 50 m elevation. The highest elevation recorded was 800 m on Slatechuck Mountain, still in forested habitat.

On 13 occasions the observer provided some information on ermine foods. There were 3 records of ermine killing domestic chickens and 3 additional cases where ermine were hunting chickens. Ermine were observed chasing rats or deer mice, and having killed swallows (*Hirundo* sp.) on 1 occasion each. Two ermine were observed together eating a muskrat. Scavenging included stealing a lunch bag from a vehicle, food from a cook house, and bait from a trap set.

Live Captures

Two ermine were captured alive, 1 in riparian forest and 1 in upland second-growth forest (Table 1). The capture site in riparian forest (lower Sachs Creek) was 700 m from the ocean and 47 m from the creek. This was a second-growth forest dominated by Sitka spruce, (10–109 cm DBH [diameter at breast height]) with substantial hemlock (31–52 cm DBH) and alder (15–41 cm DBH) in the canopy. Canopy closure was 75–80%, and there was little understory or forest floor vegetation, but substantial coarse woody debris (25% cover of pieces >5 cm diameter). Black bears were actively feeding on spawning chum salmon (*Oncorhynchus keta*) in the vicinity.

The capture site in upland second-growth forest (near mile 1 on the South Main road) was about 25 m from a swampy depression. The 30- to 40-year-old forest was dominated by Sitka spruce (17–51 cm DBH), with substantial composition of hemlock (13–36 cm DBH), redcedar (14–48 cm DBH), and alder (15 cm DBH). Canopy closure was 70–80%, and there was a moderate cover of various shrubs and ferns. Black bear sign was evident.

Deer mice were frequently caught in Sherman traps (Table 1). In 1993, deer mice were apparently more abundant in marine foreshore and marsh/fen habitats, perhaps reflecting good food sources from graminoids in these sites. The 1997 data are not directly comparable because the number of traps at each station was increased.

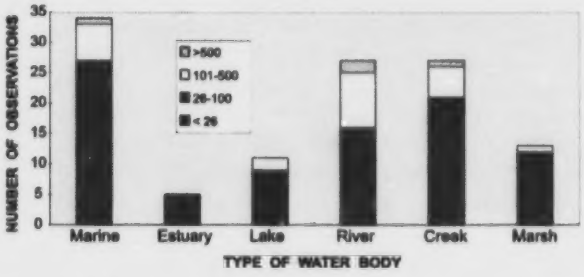

Figure 2. Frequencies of distances (m) of ermine observations from water for each of 6 water types.

DISCUSSION

Distribution

This study confirms the continued existence in the 1990s of ermine populations on both Graham and Moresby islands, and possibly on Louise Island. Ermine have colonized at least 1 other middle-sized island, Burnaby. However, they probably have little ability to cross open water except on floating logs or debris, and sufficient numbers to establish a breeding population would rarely colonize islands concurrently. Consequently, their distribution on other islands is possible but not highly probable.

The records suggest a distribution primarily in the lower elevation CWHwh1, close to the ocean or along valley bottoms. This pattern is explained in part by the distribution of humans in time and space: most settlements are at low elevation in the CWHwh1, and most major roads follow ocean front or valley bottoms. The relative scarcity of observations at low elevations on the west side of the archipelago (CWHvh1) probably results from relatively low human visitation. However, there is also evidence that this observed distribution is a fairly accurate reflection of ermine distribution: even along major roads, sightings tend to be clumped at certain water courses or river mouths, and there were very few records from upland sites in the CWHwh1 and CWHwh2 despite an extensive network of logging roads.

Population Limitation

The QCI ermine remains a very rare animal as previously indicated by Foster (1965) and Cowan (1989). The very low frequency of records in the past decade confirms this, and is noteworthy because the ermine is not a particularly secretive or inconspicuous animal (Fagerstone 1987). Evidently there are strong and persistent limitations to its potential population growth on QCI.

The ermine has a wide distribution in boreal, montane, and coastal forested habitats, and tundra environments of North America and Eurasia (Fagerstone 1987). Two principal factors act proximally to limit population growth in continental systems: access to arvicoline rodent prey, and ability to avoid predation by raptors and larger mammalian carnivores. The ermine is considered a specialist predator on arvicoline rodents, in particular those of the genera *Microtus*, *Lemmus*, *Dicrostonyx*, and *Clethrionomys*. This is based on its morphological adaptations and on the significantly enhanced survival, reproduction, and population density of ermine when these prey species are most abundant, such as during cyclic highs (Fitzgerald 1977, Simms 1979*a*, Erlinge 1981, Fagerstone 1987). However, populations can persist when arvicoline rodents are scarce, because some individuals survive on alternate prey such as other cricetid rodents, small lagomorphs, and small birds and their eggs, and by scavenging (King 1985, Fagerstone 1987). Female ermine reproduction can be significantly reduced when the system includes a variety of avian and mammalian competitors for the arvicoline prey base (Erlinge 1983). Many microtine rodents tend to prefer earlier successional plant communities, such as grassland, marsh, tundra, and shrub communities, and this likely explains the ermine's frequent preference for these habitats compared to more mature forest (Simms 1979*a*, Fagerstone 1987, Samson and Raymond 1998).

Ermine can fall prey to a variety of nocturnal and diurnal raptors, such as the snowy owl (*Nyctaea scandiaca*), rough-legged hawk (*Buteo lagopus*), and goshawk (*Accipiter gentilis*), and mammalian carnivores such as the domestic cat, red fox (*Vulpes vulpes*), and marten (Errington 1967, Weckwerth and Hawley 1962, Powell 1973, Fagerstone 1987). The recovery of red fox populations in sand dune habitats of coastal Netherlands, with a limited prey base not including arvicoline rodents, may even have caused the extinction of the local ermine population as a result of fox predation on ermine (Mulder 1990). Ermine apparently have adapted to reduce the predation risk. Their white winter pelage in northern latitudes generally makes them more cryptic, and the black-tipped tail may displace predator strikes from other parts of the body (Powell 1973, 1982). Their more intensive use of habitats with relatively heavy ground cover of coarse woody debris, herbaceous vegetation, and shrub cover (Doyle 1990, Samson and Raymond 1998) may also reflect their need to minimize predation risk.

The ermine has not been intensively studied in coastal and montane ecosystems. One study in the Cascade range of Oregon (Doyle 1990) can be considered most comparable to the situation on the QCI. The ermine, and various small rodents comprising potential ermine prey (i.e., the deer mouse and the creeping vole [*Microtus oregoni*]), were more abundant in riparian habitats than in upland habitats. This difference in abundance was correlated with significantly higher cover of deciduous and evergreen herbs and shrubs in the riparian sites compared to upland sites (Doyle 1990).

These patterns in continental ermine ecology explain a great deal of the rarity of QCI ermine, which evidently are strongly food-limited and subject to substantial predation risk. Regarding food limitation, ecosystems on QCI lack many of the ermine's preferred food species, and the ermine's access to other potential foods is subject to substantial competition. The endemic small mammal prey base on QCI is limited to the 2 species of *Peromyscus* and 1 shrew (Foster 1965, Cowan 1989), and includes none of the arvicoline rodents to which ermine are superbly adapted as predators. Consequently ermine are forced to find alternative prey. Introduced black rats are potential ermine prey (Fagerstone 1987), and are apparently hunted by ermine on QCI judging by 1 of our records. However, Cowan (1989) raises the possibility that rats may have reduced or eradicated *Peromyscus* populations, at least on Langara Island. If

this is true, even locally, on Graham or Moresby islands, the net effect of rats on ermine food availability may be negative. The introduced muskrat is unlikely to be available to ermine most of the time, because of its size and aquatic habits, and the reported case of ermine eating muskrat may have been a case of scavenging. In the relative absence of their preferred small mammal prey base, QCI ermine would be forced to broaden their prey base and include scavenged food. Foods likely include eggs, nestlings, and perhaps adults of some birds, intertidal organisms, insects, and scavenged foods including any items, such as salmon, left by larger carnivores. Some of our new records substantiate these possibilities, with a number of cases of scavenging and predation on domesticated birds.

It is unlikely that these alternative foods can compensate for the lack of arvicoline prey. This is because ermine are not well adapted to hunting many of the alternatives (e.g., most birds), because the alternatives are only locally or seasonally available (e.g. spawning salmon, rats), and the alternatives can be common in the diet of generalist ermine competitors, such as marten, red squirrels, and raccoons. Raccoons are likely exploitative competitors in that they frequently predate bird nests and feed on invertebrates in the intertidal and freshwater systems (Sanderson 1987). Red squirrels are also potential exploitative competitors, feeding seasonally on eggs and nestlings. Marten on QCI have a diverse winter diet, including many of the ermine's likely foods, in the following frequencies of occurrence: deer (35.1%); small mammals (14.4%); birds (54.6%); fish (26.8%); invertebrates (26.8%; Nagorsen et al. 1991). Nagorsen et al. (1991) found that a substantial portion of the invertebrates were marine, and believed that much of the deer and fish was scavenged. Marten are likely exploitative and interference competitors with ermine. There was general agreement among the interviewed trappers that marten populations had increased on QCI by a factor of 5–10 since the 1940s. Some attributed this increase to the introduction of the red squirrel in 1947, and others to the increase in younger seral stage forests with good habitat for deer mice. Given the frequency of deer in the marten's winter diet (Nagorsen et al. 1991), the proliferation of deer following their introduction in the early 1900s (Cowan 1989) may also have facilitated marten population growth. The declining frequency of ermine records from the 1930s and 1940s to the 1950s and 1960s may reflect a declining ermine population resulting from reduced abundance of and access to food in the face of marten competition, and perhaps marten predation. An alternate explanation is that trapper effort declined markedly after the 1940s. In any case, the increase in marten is very likely detrimental to ermine population persistence.

One selective advantage of sexual dimorphism in small mustelids may be the ability of conspecifics to exploit a wider prey base, thereby reducing intraspecific competition for food (Moors 1980). QCI ermine exhibit less sexual dimorphism than mainland subspecies (Foster 1965). Perhaps the lack of a diverse small mammal prey base means that selection to reduce intraspecific competition cannot be realized in terms of diet partitioning. Nevertheless, the competition likely still exists, and ermine may attempt to reduce it by other means, such as spacing mechanisms.

Regarding risk of predation, the following are possible ermine predators on QCI: goshawk, bald eagle (*Haliaeetus leucocephalus*), harrier (*Circus cyaneus*), sharp-shinned hawk (*Accipiter striatus*), red-tailed hawk (*Buteo jamaicensis*), black bear, marten, and domestic cat. This study uncovered 3 records of ermine killed by cats. Although ermine were not discovered in any of the marten stomach samples analyzed by Nagorsen et al. (1991), ermine are sometimes killed by marten (Weckwerth and Hawley 1962). Ironically the ermine's white winter pelage, an adaptation to reduce winter predation risk, may be a liability in the largely snow-free QCI winters. In order to reduce risk, ermine may be forced to forage more extensively at night, especially when far from cover, such as in the intertidal.

The patterns of ermine habitat use drawn from records in this study are best explained by the likely patterns of ermine prey abundance, and predation risk. Ermine were not often observed in habitats with little cover (e.g., beach, pasture, or fen). However they were frequently in forest or other cover within and near habitats (i.e., the marine foreshore and freshwater riparian), which would provide relatively good prey densities (i.e., deer mice, invertebrates, and birds). Their tendency to be further from lakes and marshes than from other water bodies might reflect a need to keep in cover. There is no indication that they are dependent on old-growth forests, though they likely benefit from coarse woody debris in any forest stand.

Inventory Techniques

Of the various inventory techniques we employed, only the process of soliciting reports, and following these up with interviews, proved very successful. This approach has the benefit of producing an historic as well as current distribution, and obtaining data for habitat and abundance estimates. We are confident that the great majority of new records are definitely of ermine, judging by the ability of the interviewee to identify key features of an ermine, the high frequency of ermine seen in white winter pelage, and the long experience of most trappers interviewed. A few records may be invalid, with red squirrels being the most likely species for confusion.

Live trapping and track stations were relatively unsuccessful in documenting ermine presence, and the question is whether the techniques were adequate or the lack of success actually reflects absence. King and Edgar (1977) suggest using a wooden trap of dimensions 60 X 17 X 11 cm. The Sherman traps we used were somewhat smaller, especially in length (38

X 12 X 10 cm), but the entrance was not substantially smaller. The captured ermine still had room to move around in the traps. The Sherman traps were metal, which is not as desirable as wood because of condensation and conductance of extreme temperatures (King and Edgar 1977). The metal may have deterred ermine under some circumstances. We would seriously consider making larger, wooden traps and track tunnels for future work, to eliminate all concerns.

We experimented with a variety of baits, but only raw bacon was successful. This may or may not indicate a preference for this bait. Dilks et al. (1996) found that eggs, broken and hard boiled, were the most effective bait in an ecosystem where ermine were likely feeding extensively on birds' eggs. Given the likely wide diet breadth of QCI ermine, we suspect that most baits offered would have been suitable to lure ermine into traps and track stations.

Resident North American ermine occupy home ranges in the order of 1 to at least 35 ha, and food-stressed individuals, which might well describe those on QCI, tend to have larger ranges (Fitzgerald 1977, Simms 1979*b*, Samson and Raymond 1998). We set live traps and track stations 50 m apart, so that 1–30 stations would have been placed in a hypothetical animal's range, and trap lines could have intersected the home ranges of 1–5 animals. This was done in order to increase the chances of detecting an ermine should it be present. A wider spacing of stations would not have allowed us to cover many more home ranges because most of the time is spent in walking between traps, or in travelling between trapping areas.

King and Edgar (1977) recommend tracking tunnels (50 X 8 cm) open at both ends, with a central ink pad, and tracking papers (17 X 7 cm) at each end. Our track stations were open at only 1 end, and had a similar sized tracking paper (16.5 X 9 cm). Dilks et al. (1996) found no significant difference in the number of ermine entering single-entrance baited tunnels compared to double-entrance tunnels.

Overall, we are confident that we would have caught or detected ermine if they were present at the locations sampled, and that our lack of results actually reflects absence during the sampling periods.

Management Implications

There are relatively few opportunities to improve the population viability for a species that is so intrinsically rare. The highest priority is to reduce predation by alien species, in particular domestic and feral cats and raccoons, by removing these species from wild and suburban habitats wherever possible. Secondly, we should manage likely ermine habitats to reduce predation risk. This could include maintaining or enhancing coarse woody debris in riparian and marine foreshore areas, and maintaining wave-washed logs above storm tide lines. Thirdly, management actions should attempt to enhance prey abundance and availability where possible. Opportunities are limited, and their benefits in terms of ermine population viability less clear. However, they could include attempts to enhance breeding bird and deer mouse populations in riparian areas, and attempts to control introduced rats, as they are competitors with deer mice. In general terms the former would include enhancing heterogeneity of canopy, understory, and ground cover species composition and structure. Prescriptions would vary with the target bird species involved, and the kinds of seed sources potentially available to deer mice.

ACKNOWLEDGEMENTS

This work was funded by the South Moresby Forest Replacement Account (SMFRA), with additional support from the British Columbia Conservation Foundation, the B.C. Ministry of Environment, Lands and Parks (MELP), and the B.C. Ministry of Forests. We thank G. Wiggins of SMFRA for administrative and logistic support on QCI, D. Nagorsen for providing records from the Royal British Columbia Museum and other North American museums, E. Rogers for supplying records from the Conservation Data Centre, and A. Thogersen for supplying records from the MELP Wild Fur Harvest. C. Husband, G. Husband, and T. Husband were invaluable in facilitating successful interviews. The following trappers were particularly helpful: J. Larose, J. Alton, E. Chapman, and S. Hale. Thanks to all those who responded to the request for records and provided interviews. The QCI Museum loaned an ermine specimen for displays. A. Midnight of B.C. Parks facilitated sampling in parks. The following people assisted with live trapping: J. Jones, K. McPherson, B. Nyberg, W. Harper, R. Kelly, B. Boese, and F. Reindl.

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Appendix A. Details of previous records of Queen Charlotte Islands ermine.

Year	Month	Data source[a]	Record type[b]	Island	Location
1898	Mar	CDC	TRD	Graham	Masset
1900	Jun	RBM	TRD	Moresby	Cumshewa Inlet
1900	Jun	RBM	TRD	Moresby	Cumshewa Inlet
1900	Jun	RBM	TRD	Moresby	Cumshewa Inlet
1900	Jul	RBM	TRD	Graham	Skidegate
1910	Aug	RBM	TRD	?	?
1910	Aug	RBM	TRD	?	?
1914	Mar	RBM	TRD	Graham	McClinton Bay
1914	Mar	RBM	TRD	Graham	?
1916	May	RBM	TRD	Graham	Masset
1916	May	RBM	TRD	Graham	Masset
1916	May	RBM	TRD	Graham	Masset
1916	May	RBM	TRD	Graham	Masset
1916	Aug	RBM	TRD	Graham	Masset
1940	Dec	RBM	TRD	Graham	Skidegate
1945	Jan	CDC	TRD	Moresby	Skidegate Channel
1959	Nov	RBM	TRD	Graham	Tlell
1963	Dec	CDC	TRD	Graham	Tlell
1965	Aug	RBM	VO	Moresby	Takakia Lake
1973	Nov	RBM	TRD	Graham	Tlell
1974	Feb	CDC	TRD	Graham	Yakoun River
1974	Feb	CDC	TRD	Graham	Yakoun River
1974	Jan	RBM	TRD	Graham	Tlell
1975	Nov	RBM	TRD	Graham	Kumdis Creek
1975	Dec	RBM	TRD	Graham	Kumdis Creek
1986	Jan	RBM	TRD	Moresby	Selwyn Point
1986	Feb	RBM	VO	Graham	Juskatla Inlet
1987	Sep	RBM	VO	Moresby	Sachs Creek
1987	Jan	RBM	TRD	Moresby	?
1987	Jan	RBM	TRD	Moresby	?

[a] CDC = British Columbia Conservation Data Centre; RBM = Royal British Columbia Museum.
[b] TRD = Trapped Dead; VO = Visual Observation.

Appendix B. Details of new records of Queen Charlotte Islands ermine collected during this study.

Year	Month	Record type[a]	Location (Island/Drainage)	Elev. (m)	Habitat type[b]	Forest harvest history[c]	Water type[d]	Distance to water (m)
1900	?	VO	Graham/Hancock	0	Beach	UC	MR/ES	5
1920	Fall	VO	Graham/Sewall	1	Bldg	RC	MR	55
1921	Spring	OSI	Graham/Sewall	1	Bldg	RC	MR	55
1922	April	OSI	Graham/Chinukundl	3	Bldg	OC	MR	35
1922	Summ.	VO	Graham/Sewall	1	Bldg	RC	MR	55
1922	Spring	VO	Graham/Sewall	1	Bldg	RC	MR	55
1924	Fall	VO	Graham/Sewall	1	Bldg	?	MR	55
1926	Summ.	VO	Graham/Sewall	1	Bldg	?	MR	55
1928	Dec	VO	Graham/Watun	0	PA	OC	MR	30
1928	Dec	TRD	Graham/Cape Ball	?	?	?	?	?
1928	Dec	TRD	Graham/Cape Ball	?	?	?	?	?
1928	Dec	TRD	Graham/Cape Ball	?	?	?	?	?
1928	Dec	TRD	Graham/Cape Ball	?	?	?	?	?
1928	Dec	TRD	Graham/Cape Ball	?	?	?	?	?
1928	Dec	TRD	Graham/Cape Ball	?	?	?	?	?

continued...

Appendix B. Continued.

Year	Month	Record type[a]	Location (Island/Drainage)	Elev. (m)	Habitat type[b]	Forest harvest history[c]	Water type[d]	Distance to water (m)
1928	Jul	VO	Graham/Masset	0	FE	UC	MR	50
1930	Jan	TRD	Graham/Yakoun	?	CF	UC	RI	25
1930	Jan	TRD	Graham/Yakoun	?	CF	UC	RI	25
1935	Jan	VO	Graham/Skidegate	2	PA	OC	CR	5
1936	Mar	VO	Graham/Cape Ball	60	FE	UC	RI	150
1936	Mar	VO	Repeat					
1936	Feb	TRD	Graham/Yakoun	?	CF	UC	CR	15
1936	Feb	TRD	Graham/Yakoun	?	CF	UC	CR	15
1936	Feb	OSI	Graham/Yakoun	?	CF	UC	CR	15
1939	Jun	VO	Moresby/Cumshewa	0	CF	OC	MR	?
1940	Nov	VO	Graham/Ain	0	CF	UC	MR/ES	50
1940	Sep	VO	Graham/Tlell	?	Bldg	UC	RI	8
1943	Jan	TRD	Graham/Kumdis	0	CF	UC	MR	50
1944	Jan	TRD	Graham/Kumdis	0	CF	UC	MR	50
1945	Jul	VO	Graham/Fife Pt	5	Bldg	UC	MR	150
1945	Feb	VO	Graham/Tlell	?	Bldg	UC	MA	25
1946	Dec	VO	Graham/Danube	30	Bldg	UC	LA	300
1946	Nov	OSI	Graham/Fife Pt	?	Bldg	UC	ES	?
1947	Jan	VO	Moresby/Skidegate	10	PA	OC	CR	20
1947	Jan	VO	Repeat					
1947	Jul	OSI	Graham/Fife Pt	5	Bldg	UC	MR	150
1952	Jul	VO	Moresby/Copper	5	CF	OC	MR	6
1963	Sep	VO	Graham/Cape Ball	3	PA	UC	ES / RI	10
1964	Mar	CAT	Graham/Tlell	?	?	?	?	?
1965	Fall	VO	Graham/Mayer	?	MF	UC	CR	500
1967	Feb	TRD	Graham/Masset	0	CF	UC	MR	3
1967	Feb	TRD	Graham/Masset	0	CF	UC	MR	3
1968	Fall	VO	Graham/Mayer	15	CF	UC	CR	?
1972	Aug	VO	Graham/Tarundl	0	Beach	OC	MR	3
1972	Jul	VO	Graham / Honna	0	FE	OC	MR	3
1972	Wint.	VO	Graham/Mayer	?	BO	UC	MA	30
1972	Wint.	VO	Graham/Mayer	?	BO	UC	MA	30
1972	Wint.	VO	Graham/Mayer	?	BO	UC	MA	30
1972	Wint.	VO	Graham/Mayer	?	BO	UC	MA	30
1973	Wint.	VO	Graham/Marie	?	CF	UC	LA	40
1973	Wint.	VO	Graham/Marie	?	CF	UC	LA	40
1973	Wint.	VO	Graham/Marie	?	CF	UC	LA	40
1973	Wint.	VO	Graham/Marie	?	CF	UC	LA	40
1973	Wint.	VO	Graham/Marie	?	CF	UC	LA	40
1973	Wint.	VO	Graham/Marie	?	CF	UC	LA	40
1973	Wint.	VO	Graham/Marie	?	CF	UC	LA	40
1974	Apr	VO	Moresby/Lagoon	10	CF	RC	MR	20
1974	Apr	VO	Moresby/Lagoon	0	Beach	UC	MR	5
1975	Nov	VO	Graham/Yakoun	20	CF	OC	RI	100
1975	Jan	VO	Graham/Blackwater	?	CF	OC	CR	60
1975	Jan	STR	Graham/Coho	?	CF	UC	CR	10
1975	Wint.	TRD	Graham/Mayer	15	FE	UC	RI	10
1975	Wint.	TRD	Graham/Mayer	15	FE	UC	RI	10
1978	Dec	VO	Graham/Yakoun	?	CF	OC	RI	800
1978	Aug	VO	Graham/Tlell	20	BO	UC	CR	50
1978	Spring	VO	Graham/Mayer	15	CF	UC	CR	?
1979	May	VO	Graham/Yakoun	10	CF	UC	RI	130
1980	Nov	VO	Graham/Slatechuck	0	ME	OC	MR	10

continued...

Appendix B. Continued.

Year	Month	Record type[a]	Location (Island/Drainage)	Elev. (m)	Habitat type[b]	Forest harvest history[c]	Water type[d]	Distance to water (m)
1980	Sep	VO	Moresby/Skaat	0	Beach	UC	MR	0
1980	Apr	OSI	Graham/Lawnhill	15	Bldg	OC	CR	80
1980	Jan	VO	Graham/Masset	30	CF	UC	MR	400
1981	Jan	TRD	Moresby/Alliford Bay	500	CF	OC	MR	350
1981	Jan	VO	Graham/Chinukundl	3	CF	OC	MR	30
1981	Nov	VO	Graham/Blackwater	?	CF	RC	CR	10
1982	Dec	TRD	Graham/Chown	0	CF	UC	RI	10
1982	Jan	TRD	Graham/Deep	0	MF	UC	CR	10
1982	Jan	TRD	Moresby/Alliford Bay	?	CF	OC	CR	?
1983	Dec	VO	Louise/Beatty Anch.	10	Bldg	OC	MR	50
1983	Oct	VO	Graham/Yakoun	?	CF	OC	RI	35
1983	Wint.	VO	Graham/Port Clements	1	Bldg	OC	CR	11
1984	Nov	VO	Graham/Yakoun	?	CF	OC	RI	7
1984	Nov	VO	Graham/Mayer	?	BO	UC	MA	35
1985	?	VO	Moresby/South Bay Rd	?	CF	OC	MR	500
1985	Nov	VO	Graham/Datlamen	20	CF	RC	RI	300
1985	Jan	TRD	Burnaby/Dolomite	?	CF	UC	MR	500
1986	?	VO	Graham/Yakoun	?	CF	UC	CR	?
1986	Jul	Vo	Moresby/South Bay Rd	75	CF	RC	MA	350
1986	Jul	VO	Louise/Mathers	1500	CF	RC	MR	1600
1986	Jul	VO	Repeat					
1986	Jan	VO	Graham/Miller	5	MF	?	MR	30
1986	Jan	TRD	Moresby/Mosquito	?	CF	OC	LA	300
1986	Feb	VO	Graham/Mayer	?	BO	UC	MA	30
1986	Feb	VO	Graham/Mayer	?	BO	UC	MA	30
1987	Dec	VO	Graham/Yakoun	?	CF	OC	RI	5
1987	Nov	VO	Graham/Yakoun	10	CF	UC	RI	75
1987	Sep	VO	Graham/Brent	250	CF	OC	CR	1
1987	Jun	VO	Graham/Brent	0	Bldg	UC	MR	15
1987	Feb	VO	Graham/Phantom	?	CF	UC	CR	75
1988	Jul	VO	Graham/Rose Spit	0	Beach	UC	MR	70
1988	Jul	VO	Repeat					
1989	Dec	VO	Graham/Florence	?	CF	RC	?	1000
1989	Sep	VO	Graham/Tlell	?	CF	UC	MA	20
1989	Jul	VO	Graham/Gold	?	CF	RC	RI	4
1989	Feb	VO	Graham/Kumdis	15	CF	OC	CR	100
1991	?	VO	Graham/Kagan	800	CF	UC	LA	50
1991	Nov	VO	Moresby/Sandspit	1	Beach	OC	MR	30
1991	Jul	VO	Graham/Mayer	?	CF	UC	?	?
1991	Jul	CAT	Graham/Tlell	?	?	?	?	?
1991	Jun	VO	Graham/Watun	0	CF	UC	ES / RI	5
1991	Feb	STR	Moresby/Mosquito	10	CF	OC	LA	80
1991	Jan	TRD	Moresby/Mosquito	?	CF	OC	MR /CR	?
1991	Sep	VO	Graham/Tian Head	0	CF	UC	MR	5
1992	Dec	VO	Graham/Yakoun	?	CF	OC	RI	200
1992	Dec	STR	Moresby/Chroustcheff	5	PA	OC	MR	?
1992	Jul	VO	Graham/Phantom	?	CF	RC	RI	1000
1992	Jul	VO	Graham/Yakoun	?	CF	RC	?	?
1992	Mar	VO	Moresby/Copper	?	MF	OC	MR	21
1992	Feb	VO	Graham/Miller	50	MF	UC	CR	60
1992	Jan	STR	Graham/Sue	?	CF	RC /OC	CR	?
1993	Dec	VO	Graham/Yakoun	?	SII	RC	RI	50
1993	Oct	VO	Graham/Mamin	?	CF	RC	CR	200

continued...

Appendix B. Continued.

Year	Month	Record type[a]	Location (Island/Drainage)	Elev. (m)	Habitat type[b]	Forest harvest history[c]	Water type[d]	Distance to water (m)
1993	Sep	VO	Louise/Skedans	1	CF	OC	MR	5
1993	Jul	VO	Moresby/Copper	1	Beach	OC	MR	10
1993	Jun	VO	Graham/Tlell	3	DF	OC	MR	25
1993	Apr	VO	Graham/Lawn Pt	1	MF	OC	MR	40
1993	Feb	VO	Graham/Yakoun	?	CF	OC	RI	200
1993	Jan	VO	Graham/Gregory	5	CF	OC	RI	200
1993	Nov	VO	Graham/Mayer	?	CF	UC	MA	30
1994	Dec	VO	Moresby/Sachs	15	CF	OC	MR/CR	200
1994	Jul	VO	Graham/Riley	?	CF	UC	?	?
1994	Mar	VO	Graham/QC City	8	CF	OC	MR	500
1994	Feb	STR	Graham/Survey	?	CF	RC	CR	100
1994	Jan	VO	Graham/Yakoun	?	CF	UC	CR	100
1995	Oct	VO	Graham/Towhill	10	MF	OC	RI	200
1995	Sep	VO	Graham/Yakoun	?	CF	UC	RI	20
1995	Jul	VO	Graham/Rennell	5	CF	OC	MR	25
1995	Mar	VO	Graham/Mamin	400	CF	UC/RC	CR	50
1995	Jan	STR	Graham/Survey	?	CF	OC	CR	80
1996	Dec	STR	Moresby/Deena	1	CF	OC	MR	10
1996	Dec	VO	Graham/Kumdis	50	MF	UC	CR	300
1996	Oct	VO	Graham/Davidson	?	CF	RC	CR	12
1996	Sep	VO	Graham/Delkatla	0	CF	OC	ES	10
1996	Sep	VO	Repeat					
1996	Jul	VO	Graham/Yakoun	250	CF	OC	RI	50
1996	Mar	VO	Graham/Yakoun	?	CF	OC	CR/RI	20
1996	Feb	TRD	Graham/Honna	?	?	?	?	?
1996	Feb	VO	Graham/Yakoun	?	CF	OC	CR	10
1996	Feb	VO	Graham/Yakoun	?	CF	OC	CR	10
1996	Jan	VO	Graham/Mayer	50	BO	UC	MA	15
1997	Aug	VO	Graham/Masset	10	MF	OC	CR	250
1997	Aug	VO	Moresby/Sachs	1	CF	OC	MR/CR	0
1997	Aug	VO	Graham/Kumdis	50	CF	UC/RC	MA	100
1997	Aug	VO	Moresby/Blaine	10	CF	OC	MR	240
1997	Aug	VO	Moresby/Heather	20	CF	RC	CR	?
1997	Jul	VO	Graham/Lawnhill	3	MF	UC	MR	30
1997	Jun	VO	Moresby/Government	?	CF	UC	RI	0
1997	May	VO	Graham/Towhill Rd	0	MF	OC	MA	60
1997	Feb	TRD	Graham/Honna	200	CF	OC	LA	500
1997	Aug	CAT	Graham/QC City	?	?	?	?	600
1997	July	VO	Graham/Lawnhill	10	CF	UC	CR	?
1997	Nov	VO	Graham/Mamin	30	CF	RC	RI	200
1997	Nov	VO	Graham/Mamin	20	CF	OC	RI	500

[a] VO = Visual observation; TRD = Trapped dead; STR = Snow track; CAT = Killed by cat; OSI = Other sign.

[b] CF = Coniferous forest; MF = Mixed forest; DF = Deciduous forest; SH = Shrub; FE = Fen; PA = Pasture; BO = Bog; Bldg = Building.

[c] UC = Uncut; OC = Old clearcut (>10 yr); RC = Recent clearcut (<10 yr).

[d] MR = Marine; ES = Estuary; RI = River; CR = Creek; LA = Lake; MA = Marsh.

The Tiger Salamander in British Columbia: An Amphibian in an Endangered Desert Environment

John S. Richardson
Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
and Department of Forest Sciences
2424 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada
jrichard@interchg.ubc.ca

Walter Klenner
Research Section, British Columbia Ministry of Forests
515 Columbia Street
Kamloops, BC, V2C 2T7, Canada

Jeff Shatford
Department of Forest Sciences
2424 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

The Red-listed tiger salamander (*Ambystoma tigrinum melanostictum*) is found in a limited range in the South Okanagan, where it is subject to large-scale habitat alienation. We studied a series of 10 ponds to determine larval growth rates, juvenile recruitment, movement of adults to and from breeding ponds, and terrestrial habitat use by adults during the summer. Ponds varied in the rate of larval growth, size at metamorphosis and timing, and the number of new recruits. Two of the most productive ponds were shallow and most susceptible to drying prior to larval development, indicating that the productivity of a given pond may vary from year to year. Through the summer months adult tiger salamanders did not move >150 m from the pond where they were captured, and used open sagebrush-grassland habitats. In this arid environment they did not select dense cover or mesic sites, and likely avoided dehydration by entering small mammal burrows.

Key words: *Ambystoma tigrinum*, amphibian, grasslands, Okanagan, populations, radiotelemetry, tiger salamander, wetlands.

Amphibians are sensitive to land-use alteration and there is widespread concern that environmental contamination and land conversion are responsible for their decline. The tiger salamander (*Ambystoma tigrinum melanostictum*) is a widely distributed species in North America, but within British Columbia it is only found at a limited number of sites in the South Okanagan and Similkameen areas (Sarell 1996). The species is Red-listed by the Wildlife Branch of the British Columbia Ministry of Environment, Lands and Parks (1996). Habitat protection for this species and other organisms in grassland and sagebrush areas is urgent because of the rapid alienation of the remaining habitat. Despite its broad distribution in North America, there have been no studies on the species in the western part of the continent, and none on habitat requirements in a landscape as xeric as the bunchgrass biogeoclimatic zone in British Columbia (Lloyd et al. 1990).

Most amphibians have a complex life history with dependence upon 2 or more habitats to complete their life cycles: an aquatic breeding and larval habitat; and a terrestrial habitat, where juveniles and adults spend most of their time. In addition, amphibian adults need appropriate winter resting places to avoid freezing and predation. In setting priorities for land conservation for particular species, it is essential that the full needs of the species be evaluated. In this study we had several objectives. The first was to evaluate the productive capability of a series of ponds and wetlands by monitoring larval densities, growth, and survival to age of metamorphosis. We also estimated the numbers and directions of dispersal of newly transformed juveniles from many of our study ponds. Finally, we began an evaluation of terrestrial habitat use by adult salamanders following the spring breeding period by tracing the movements of adults with radiotelemetry implants. The results presented here are part of the ongoing study of a population of tiger salamanders in southern British Columbia.

STUDY AREA AND METHODS

We studied tiger salamanders at 10 wetlands or ponds around and including White Lake (49°18'N, 119°37'W),

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

southwest of Penticton, B.C. The west side of White Lake has been purchased by The Nature Trust of British Columbia and the remainder of the study area is part of a Government of Canada grazing lease administered by the Dominion Observatory. The 10 ponds vary in depth and area, as well as in the vegetation of the area surrounding each, despite being in an area of only approximately 10 km^2 (Fig. 1). The ponds also vary in the permanence of standing water, and some ponds may dry out, or nearly so, during some years, depending on weather and depth. This variation between ponds provides a convenient contrast for assessing the population ecology and habitat use of tiger salamanders in relation to changing habitats in space and time.

Larvae were trapped in all 10 ponds with baited (using commercial pet food in plastic canisters) minnow traps secured near the water surface during several sampling sessions in 1997 and 1998. Larvae were measured for total length and weighed. A small notch was cut in the tail fin prior to their release at the point of capture, thereby preventing the double-counting of an individual in a sampling bout of several days. We assumed changes in mean size between dates within a pond were due only to growth and not to any size-specific mortality.

Arrays of drift fences and pitfall traps were established to capture newly transformed juveniles leaving ponds. Pond 2 was completely encircled and all metamorphosed individuals could be captured, while pond 6 was also completely encircled, but only completed after most of the larvae had transformed. At another 3 ponds (White, 3, and 4) we established 4 linear drift fences of 20 m each, and pitfall traps at each end and in the middle. We used capture rates at these linear fences to estimate the number of newly metamorphosed juveniles leaving these ponds and to record the phenology of transformation. Each newly transformed juvenile received a pond-specific toe clip for identification on future capture. Samples of these animals were measured and weighed.

In spring of 1998 we used pitfall arrays to capture adults returning to ponds to breed. Adults were measured and weighed, and 14 animals were captured for surgical implantation of radio-transmitters (Holohil BD-2GH, 1.6 g) as they left the ponds. After surgery, these individuals were allowed to heal for about a week (until the incision had healed). The animals were released at the point of capture and tracked by radiotelemetry for up to 4 months.

RESULTS

Of the 10 ponds studied, tiger salamander larvae were found in 8: White Lake, 2, 3, 4, 5, 6, 7, and 10. Larvae were never found in ponds 8 and 9. We captured larvae at pond 4 in 1997, but not in 1998. Ponds 2 and 5 both dried early in 1998 (by 31 July) and apparently no larvae survived. The trapping success in ponds 4 and 7 (ponds that had resident turtles) was variable, and very low in comparison with the other ponds.

Larval growth rates varied considerably among ponds, as did the apparent timing of hatching and completion of growth. In both years the total growth of larvae was highest in White Lake. Larvae in White Lake reached the size for metamorphosis earlier on average than populations in the other ponds. Larval size in White Lake was significantly higher than in other ponds by mid-summer in each year

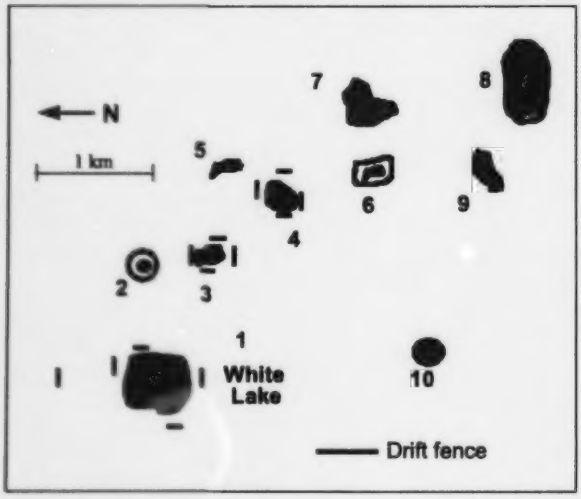

Figure 1. Map of the study area indicating the relative size and position of the 10 ponds studied in the White Lake area.

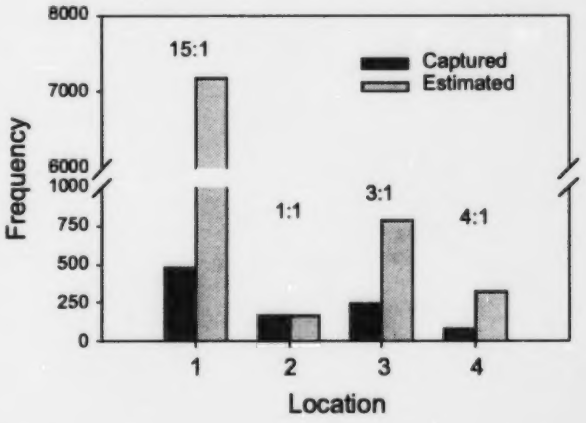

Figure 2. Actual and estimated numbers of newly recruited juveniles from 4 of the ponds, based on pitfall trap counts between mid-August and early October. Extrapolations to total numbers were based on the simple ratio of drift fence length to pond perimeter.

(ANOVA, $p < 0.0001$ for both years). In 1997, pond 10 larvae were intermediate in size between White Lake individuals and those from other ponds ($p < 0.05$, Tukey's test). Although we began larval trapping later in 1997 than in 1998, it was apparent that there were higher densities and that they stayed in the ponds longer in 1997 than in 1998. In 1997 the larval density was sufficiently high along the margins of White Lake that the water would "boil" when disturbed by our activities as we walked along the perimeter, but this was never observed in 1998. In 1998 larval densities were much lower than in the previous year, and the density of larvae appeared to decline as early as mid-July.

Newly transformed individuals were trapped in 1997 at our arrays of pitfall traps and drift fences. White Lake was by far the biggest contributor of new recruits to the population, with an estimate of almost 7,200 new juveniles in 1997 based on our array of drift fences (Fig. 2). The other 3 ponds where emigrating juveniles were censused (pond 6 drift fences were not in place for the whole period) contributed modest numbers to the overall population, but in combination only about 18% of the numbers from White Lake (Fig. 2). Juveniles from White Lake were about 15% heavier on average than those from pond 6, which had the next largest metamorphs (Fig. 3A). There were significant differences in size at transformation among the 5 ponds where we operated pitfall traps (ANOVA, $p < 0.0001$). The coefficients of variation were between 18.6 and 31.9% across the 5 ponds and, in part, reflect the inclusion of both sexes and individuals across the period of transformation (it is a well-known phenomenon that size at metamorphosis declines as the season proceeds). Ponds 2 and 3 had significantly smaller metamorphs (Tukey's test, $p < 0.05$), despite having high numbers of recruits relative to ponds 4 and 6.

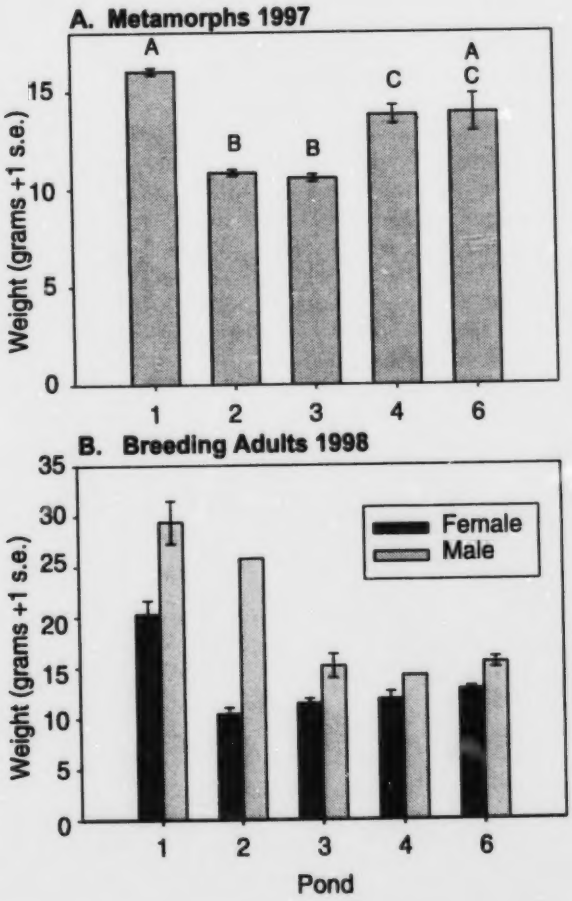

Figure 3. Weights of (A) newly transformed juveniles from late summer 1997, and (B) adults returning to ponds to breed in spring 1998. Bars with different letters are significantly different ($p < 0.05$). Bars lacking standard errors (males, ponds 2 and 4) are for single individuals only.

Pond 2 was completely encircled by a drift fence, so we were able to plot the direction of emergence of newly transformed juveniles in 1997. There was a clear pattern of tiger salamanders emerging from the southwest side of the pond (Fig. 4). The direction was towards the sagebrush grasslands and away from any wooded areas. Long-toed salamanders (*A. macrodactylum*), which were also common in the pond, showed a clear direction to their emergence as well: to the northeast, i.e., upslope towards a wooded area.

In spring 1998 adults were captured using pitfall traps and drift fences that had been established for juveniles the previous summer. Our sampling began on 31 March, after the start of the breeding migration, and adults were last trapped on 22 May. Most adult males had entered the ponds before we started trapping. We found significant differences in the mean size of males and females overall (17.4 vs. 14.7 g for males and females respectively, 2-way ANOVA, $p < 0.0001$). There were significant differences in the mean sizes of adults returning to each pond (2-way ANOVA, $p < 0.0001$), with White Lake adults being largest (Fig. 3B, $p < 0.05$, Tukey's Test).

Of the 14 adults that were implanted with radio-transmitters, 9 were tracked between 57 and 93 days. One animal did not survive the implant procedure; 1 was killed by a raptor within 2 weeks of release; and 3 others could not be located despite extensive searching. The 9 transmitter-equipped animals were located 146 times, with observations distributed across the 2- to 3-month monitoring period. None of the animals being monitored travelled >150 m from the pond where they had been captured. Upon release, all animals moved away from the pond within a week and, with the exception of one animal, inhabited dry sagebrush-grassland habitat, often in areas with >30% mineral soil exposure. Movement patterns varied between animals, but in general

were characterized by extended periods (e.g., 2–3 weeks) of restricted movement, where activities were constrained to a 5-m radius area, punctuated by abrupt movements of 10–100 m to a new location. Salamanders were always in subterranean burrows during the day, and used the loose soil and tunnels excavated by Great Basin pocket mice (*Perognathus parvus*). One salamander was repeatedly found at the same location in an area of mesic soil within 10 m of the release site. When excavated, this animal was alive and <30 cm below the surface.

DISCUSSION

Larval habitats vary in productivity, growth rates of larvae, and size/timing of transformation. White Lake was the most productive of the set of 10 ponds, on the basis of growth rate, densities, and numbers of juveniles produced. Pond 10 was also quite productive, but as with White Lake, very shallow and prone to drying during summer prior to tiger salamander metamorphosis. Other ponds were less productive by most measures, but still contributed recruits to the population.

Juveniles leave ponds at a time of year when they are potentially vulnerable to desiccation, and vegetative cover may be critical. Most of the captures of juveniles in our pitfall trap arrays coincided with precipitation, suggesting the juveniles have some capacity to remain in refugia until weather becomes suitable. Cover provided by large pieces of wood or small mammal burrows may be critical to survival of juveniles at this stage of their lives. The observation that juveniles from pond 2 dispersed primarily in 1 direction is intriguing, suggesting there may be some cues for habitat selection. We assumed *a priori* that any directionality would be towards forest cover, which was clearly rejected by the observation of almost all juveniles moving in a direction away from any trees.

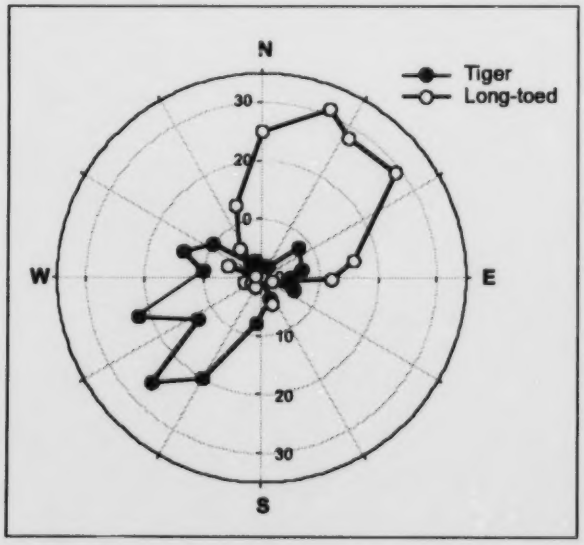

Figure 4. Frequency of juvenile tiger salamanders and long-toed salamanders (*Ambystoma macrodactylum*) captured in pitfall traps as they dispersed from pond 2 in 1997.

Adults can survive in dry sagebrush-bunchgrass habitats, even during periods of high summer temperatures (>35°C) and extended periods without precipitation. Fine-scale habitat selection, for example, for areas with dense vegetation or low mineral soil exposure, or for mesic microsites as indicated by shrubs (e.g., *Amelanchier alnifolia*), was not observed. These observations are very preliminary, but may indicate a broad habitat tolerance by terrestrial adults in summer.

Studies of movement patterns of other *Ambystoma* salamanders showed that adults rarely moved far from ponds, although individuals varied by using either forested or open sections with little overlap in habitat use between individuals (Madison and Farrand 1998). Small mammal burrows were used by *A. maculatum* both during the summer (mostly shrew [*Sorex* spp.] burrows) and winter (primarily deer mouse [*Peromyscus maniculatus*] burrows) and served as terrestrial refuges (Madison 1997). In our study area, adult and juvenile tiger salamanders used Great Basin pocket mouse burrows. We do not know where the adults and juveniles overwinter, but winter habitat is probably a critical resource for these populations.

The capriciousness of weather between years means that permanence of water bodies may vary and create high between-year variability in larval survival rates. Data for water levels and duration in White Lake do not exist, but nearby Mahoney Lake (about 10 km ESE from White Lake) had declining water levels during 1985–1995 (Northcote and Hall submitted). White Lake and many of the surrounding ponds are quite shallow, and even in the relatively wet years of 1997 and 1998, some of the ponds dried completely before tiger salamander larvae could transform. White Lake frequently dries out completely by late July (R. [Dick] Cannings, Honorary Curator, Vertebrate Museum, University of British Columbia Zoology Department, pers. comm.), and even in a wetter year may diminish in size prior to larval metamorphosis, potentially limiting the production of new recruits. Prolonged periods when the shallow ponds contribute no recruits to the adult population may lead to large declines in numbers depending upon success rates in the deeper ponds. Based on our results White Lake is clearly the biggest contributor to this metapopulation; however, we do not know the degree of fidelity individuals show to their natal pond. The similar rankings of sizes of adults returning to ponds with the size of metamorphs leaving suggests that there may be a high degree of philopatry. The potential for periodic increases in populations based on recruitment from White Lake may be

short-lived, depending on longevity of adults. In a year, or many years in sequence, when White Lake produces no recruits, the deeper ponds in the area may be the only source of new recruits. This reinforces the importance of alternate habitats for population or metapopulation conservation.

There are still many critical questions about the population ecology of this species for which we have no answers. Amphibians are dependent upon suitable wintering habitats where they can escape the lethal freezing conditions. We do not know where tiger salamanders in the population we studied find such hibernacula. Predators in the more permanent ponds may make these less suitable habitat for tiger salamander larvae, but the role of predators is still unknown. These ponds may act as individual populations with high degrees of philopatry or, conversely, interannual dispersal between ponds may mean this is a large population with multiple breeding sites. The degree to which each pond is independent of the others will affect how critical each is as a habitat to be conserved. The potential for long-distance dispersal depends upon corridors of suitable and continuous habitat. We do not know the capacity for dispersal, nor the effect of barriers (partial or complete) such as roads and agricultural developments for tiger salamanders in this arid region of North America.

MANAGEMENT IMPLICATIONS

The series of ponds we studied near White Lake provide habitats of variable value for the tiger salamander. A series of larval habitats (shallow and deep ponds) seems critical to the maintenance of this population. Terrestrial adults and juveniles can manage to survive in relatively open habitat and our preliminary findings suggest terrestrial adults have broad habitat tolerance. Reproduction and productivity in drought years, the role of introduced species (e.g., rainbow trout), and the importance of overwintering habitat need to be addressed before informed management plans are developed for this species.

ACKNOWLEDGEMENTS

This research was supported by the Habitat Conservation Trust Fund, the British Columbia Ministry of Forests, and the University of British Columbia. We thank T. Baker, C. Dyck, and C. Strong for help with data collection, and we appreciate the cooperation of the land-owners and holders of the grazing tenures for permission to work in the area around White Lake.

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Status of the Northern Leopard Frog in the Creston Valley Wildlife Management Area, British Columbia

Heather L. Waye[1]
Manning, Cooper and Associates
1278 Laurel Road, R.R. 3, Sidney, BC, V8L 5K8, Canada

Key words: Creston Valley, northern leopard frog, *Rana pipiens*.

EXPANDED ABSTRACT

The northern leopard frog (*Rana pipiens*) occurs extensively throughout eastern North America, but has been more limited historically in the west. From the mid-1960s to mid-1980s there was a general decline of leopard frogs over much of their range. Sharp declines were noted in the late 1970s in the western provinces and states, with complete loss of the species at some sites. This species used to be found throughout the southern half of Alberta, but now there are apparently only 27 breeding locations. They are now limited to a small area of central Washington, and are gone from much of their range in Idaho. However, they seem to have made a minor comeback over the last few years in Alberta, Saskatchewan, and Idaho.

The records in British Columbia have been few and far between, with only about 12 historic locations in the Okanagan, Kootenay, and Columbia river valleys, and 1 introduced population on Vancouver Island. This species declined in British Columbia in the late 1970s, and by the mid-1990s was only known to occur near the town of Creston, south of Kootenay Lake. The northern leopard frog has been placed on the provincial Red List, and the British Columbia population is now considered to be endangered by COSEWIC (the Committee on the Status of Endangered Wildlife in Canada).

The Columbia Basin Fish and Wildlife Compensation Program (CBFWCP) funded a pilot project in 1996 to determine the status of the northern leopard frog in the Creston Valley Wildlife Management Area. This study found 3 males calling at 1 breeding site, and 4 frogs throughout the summer. In 1997, the CBFWCP began funding a 3-year project with the following objectives:

1. determine the distribution and abundance of leopard frogs in the management area;
2. establish whether this population is breeding successfully, and what habitat they require to do so;
3. identify and describe overwintering sites; and
4. provide management suggestions based on the information collected.

We are now at the start of the third year of this study.

We split the field days between the spring, summer, and fall active seasons. In spring, we did calling surveys to find out what habitat the frogs preferred for breeding, and to count the number of mature males. We did visual encounter surveys in all the sessions, to look for egg masses and tadpoles in the spring, metamorphic juveniles in the summer, and adults all year. When we caught frogs, we weighed and measured them, tried to determine their sex, and drew their spots for future identification. We also recorded information about the surrounding habitat, such as plant species, plant heights, temperature, and distance to water. In 1997 we also used minnow traps, and pitfall and drift-fence arrays, but they were not very successful for the effort required. It turned out that the design of the pitfalls allowed the frogs to escape, and the amount of time it took to install and check them was better spent walking around and looking for frogs. In the fall of 1998 we used radiotelemetry to track some of the frogs.

We caught 37 frogs in 1997—more than we expected, based on the number seen in 1996—and we caught another 116 frogs in 1998. We did not sample in the summer of 1998 so we could spend more time looking for frogs and tracking radio-tagged frogs in the fall. There did appear to be more frogs around in 1998, but we had also become better at finding frogs and were spending more time in the field looking for them. However, we did catch more frogs per unit effort (0.07 new frogs/person-hour in 1997, 0.17 in 1998). The recapture rate of marked frogs was 3% from 1997 to 1998 (only 1 frog marked in 1997 was recaptured in 1998). Low recapture rates are not unusual in herpetological studies, and we only marked 37 frogs in the first year, so this low recapture rate is not entirely unexpected. We tended to see the same frogs from 1 day to the next, and we would see individuals

[1] Present address: c/o Department of Zoology, 3029 Cordley Hall, Oregon State University, Corvallis, OR 97331, USA
wayeh@bcc.orst.edu

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

again a few weeks later, so we feel that the act of catching the frogs did not affect their survival.

We found calling males at 2 locations in the marsh in both years, and in 1997 we found an egg mass in the spring, metamorphic juveniles in the summer, and young of the year in the fall. In 1998 we found a pair of adults in amplexus, 3 egg masses, tadpoles in the spring, and young of the year in the fall. The egg masses and tadpoles were found at 1 of the breeding sites only, so we do not have evidence of successful breeding at the other calling site. There were definitely more males at the breeding sites in 1998 (6 captured in 1997, 29 in 1998), although we were at the site at slightly different times each spring and so might have missed the bulk of the calling males in 1997.

Most of the males at the calling sites in 1998 were much smaller than the males in 1997. They were probably born in 1996 and were breeding for the first time, and many of the larger males from 1997 likely did not survive through the year to breed again in 1998. It is possible that the generations in the population are barely overlapping, and we are seeing a turnover in the population from mostly older to mostly younger frogs. The young of the year in the fall of 1998 were much larger than those in 1997. This is probably because the breeding was earlier in 1998 and the summer temperatures were hotter, giving the juveniles more time to grow before the fall. However, we found proportionately fewer frogs that were obviously young of the year in 1998.

We attached radios to 14 frogs in the fall of 1998 (no more than 6 frogs at one time). We recorded a total of 189 frog-days (i.e., number of days a frog carried a radio added up for all frogs) and 110 locations, which gave us some interesting information on the habitats the frogs use in the fall. For example, 1 frog moved through a stand of cattails (*Typhus latifolia*) and mud that we did not think frogs would use at all, while another was hiding under the roots of a bunch of cattails. A frog would often stay in 1 location for days, move 5–50 m away for a few days, then move back.

Over all, the frogs ended up at the edges of bodies of deeper water (water that deepened to over 1 metre), and were still there at the end of the study at the end of October. On cold mornings we would find some of the frogs completely submerged in the water. Leopard frogs are known to hibernate underwater on top of the mud, and the frogs in Creston appeared to prefer water with vegetation along the bottom, with little or no current. The bodies of water appeared to be of sufficient volume to provide enough oxygenated water to the frogs over the winter. Three of the frogs eventually went down small mammal holes dug into the banks along the edge of the water, and were at least 10 cm below the surface. On warm days, we could find the frogs on the surface, but later in the fall, as the morning temperatures reached freezing, the frogs appeared to stay underground all day.

The only known population of northern leopard frogs in British Columbia is in an area managed for many species, some of which are also Red-listed. A balancing act is required to maintain the area to benefit all, and requires good knowledge of the requirements of those species. We know that the breeding habitats of leopard frogs are rare in the Creston Valley. They appear to need beds of spikerush (*Eleocharis palustris*) in shallow, open water, and the 2 sites where we have observed breeding activity are the only locations in the valley where we have found this habitat. Therefore, we need to know how to protect, enhance, and possibly create this type of habitat. Overwintering sites are also critical, and although the leopard frogs seem to be able to find the conditions they require in a variety of habitats, these sites must be considered in management practices. There are possibly 2 subpopulations in this marsh, 1 concentrated around each of the calling sites, and this must be determined, as it will affect how the population is managed. We need more information about the population size, as only 2 years of captures does not tell us whether the population is increasing or decreasing. We need to determine summer habitat use and spring daytime habitats, so we plan to continue with the radiotelemetry this year. There are ongoing searches in the Columbia Valley for other populations of leopard frogs and, while these have not been successful, there are many locations that have not yet been searched. We hope to continue public education about this species and encourage members of the public to report any likely-looking frogs to the British Columbia Ministry of Environment, Lands and Parks or the Conservation Data Centre.

Pursuing the Unseen: Habitat Use by Rubber Boas in Creston

R. St. Clair

Department of Biology, University of Alberta, Edmonton, AB, T6G 2E9, Canada
rstclair@ualberta.ca

ABSTRACT

The protection of animals of which we know little is one of the onerous tasks facing wildlife managers. Rubber boas fall into this category. Nothing has been published about their basic ecology, probably because they are burrowing and rarely encountered. Yet rubber boas are of management concern (e.g., they are "Identified Wildlife" under the Forest Practices Code) because their abundance is unknown and they are found in sensitive habitats such as temperate forests. To address this lack of knowledge, I have been studying habitat use of rubber boas in Creston, British Columbia. Last year, rubber boas were followed using temperature-sensitive telemetry. I recorded habitat characteristics including structure, thermal environment, and associated plant communities. Rubber boas spent most of their time under rocks in forest openings. By selecting cover rocks of appropriate thickness and moving vertically in the soil, they maintained favourable body temperatures while remaining hidden. The focus of work this winter is the discovery and description of denning sites. These preliminary descriptions of habitat use will be expanded to derive a predictive model of habitat requirements for this sensitive species.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Inventories of Sagebrush Northern Bog Lemming, Williamson's Sapsucker, and Mountain Beaver in the Southern Interior of British Columbia

Les W. Gyug
Okanagan Wildlife Consulting,
3130 Ensign Way, Westbank, BC, V4T 1T9, Canada
Les_Gyug@bc.sympatico.ca

Alan Peatt
British Columbia Environment
201-3547 Skaha Lake Road, Penticton, BC, V2A 7K2, Canada

ABSTRACT

Three species were inventoried in the Southern Interior of British Columbia (B.C.) from 1996 to 1998 in areas where conflicts were likely to develop with proposed timber harvesting. Sagebrush northern bog lemmings (*Synaptomys borealis artemisiae*) were found at 2 sites, and are only known from 8 localities (5 in British Columbia, 3 in Washington State). They are found mainly in fens, swamps, and marshes at 1,600 m or higher. Twenty Williamson's sapsucker nests (*Sphyrapicus thyroideus thyroideus*) were found with most between 1,000 and 1,200 m elevation east and south of Penticton, B.C. We found 65% of the nests were in large (>70 cm diameter at breast height), live western larch with dead or broken tops. Four of 10 trees for which we have multiyear data were reused for a second year of nesting. The population of Williamson's sapsuckers in British Columbia is estimated to be 100 nesting pairs. We found 634 mountain beaver (*Aplodontia rufa raineri*) nest sites on the east side of the Cascade Mountains, primarily at higher elevations in the Engelmann Spruce–Subalpine Fir moist warm subzone, along seeps and small creeks in areas of deep soil. They can live in any successional stage but we found them to be very sensitive to soil disturbance caused by heavy machinery during logging or site preparation.

Key words: *Aplodontia rufa raineri*, inventory, *Larix occidentalis*, mountain beaver, sagebrush northern bog lemming, Southern Interior, *Sphyrapicus thyroideus thyroideus*, *Synaptomys borealis artemisiae*, western larch, Williamson's sapsucker.

This project was designed to collect locality and habitat information for Red- or Blue-listed taxa in the Southern Interior of British Columbia within proposed cutting permit and timber sale areas on 1996, 1997, and 1998 forest development plans (FDPs) in the Merritt, Vernon, Penticton, and Boundary Forest Districts. Species or subspecies that were inventoried and are reported on here include sagebrush northern bog lemming (*Synaptomys borealis artemisiae*), Williamson's sapsucker (*Sphyrapicus thyroideus thyroideus*), and mountain beaver (*Aplodontia rufa raineri*). In 1997 and 1998, additional objectives were to better define the effects of timber harvest and silviculture treatments on these species so that mitigative strategies could be incorporated into timber harvesting and silviculture prescriptions for areas proposed for timber harvest.

STUDY AREA

The nominal study area included the entire Merritt, Vernon, Penticton, and Boundary Forest Districts in the Southern Interior of British Columbia. The actual areas inventoried depended on the known or suspected distribution of each taxon, and were also limited to the areas where timber harvesting was proposed on 1996, 1997, and 1998 FDPs. Before this project, the sagebrush northern bog lemming's distribution was poorly known; the only published occurrences were on the east side of the Cascade Mountains southwest of Princeton (Anderson 1932) with 1 additional occurrence about 3 km from one of Anderson's (1932) records (personal data, 1994). The mountain beaver is found in the Cascade Mountains southwest of Merritt (Cowan and Guiguet 1965) with its eastern range boundaries poorly defined (Orchard 1984). The Williamson's sapsucker has been found breeding throughout the Southern Interior (Cooper 1995). Specific areas selected for inventory of each species depended on overlap of areas proposed for timber harvest in the 1996–1998 FDPs with the actual habitat presumed to be used by each species.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

METHODS

Sagebrush Northern Bog Lemming

Only 2 areas, Paul Creek (1996) and Easygoing Creek (1997) (Fig. 1), were specifically inventoried for sagebrush northern bog lemmings. Trapping was carried out in the treeless wet subalpine meadows, fens, swamps, and marshes beside the proposed cutting permits. Three transect sampling lines (50 Victor mouse snap traps set in pairs at 10-m intervals for 2 nights) were set out in each location. Additional live and snap trapping was carried out at Paul Creek in 1997 to determine if bog lemmings made any appreciable use of the forest around the bogs, and whether timber harvest would change their habitat use. Five additional snap trap sampling lines were set in riparian forest on streams leading into the Paul Creek marshes and in the headwater marshes of nearby Smith Creek. Live trapping (4 x 7 trap grid at 20-m intervals) with Sherman live traps was undertaken; each grid was centered on the forest/marsh interface where bog lemmings were found in 1996. Traps were pre-baited for up to 3 weeks before use and then set open for 2-night trapping sessions in September 1997 for a total of 672 trap nights and 1,008 trap checks.

Williamson's Sapsucker

Tapes of the calls and drums of Williamson's sapsuckers were played to solicit responses from early May through early June in 60 areas selected for inventory (18 in 1996, 30 in 1997, and 12 in 1998). Where Williamson's sapsuckers were detected, we returned in late June to thoroughly search for nests by listening for begging calls of young and by looking and listening for adults foraging, drumming, or calling near the nest. Data were collected on nest trees with follow-up visits made to each nest in subsequent years.

To estimate density of nesting pairs in close to "ideal" habitat, complete inventories of nests were attempted in late June 1997 and 1998 in Woodlot 411-1 at Johnstone Creek. These estimated densities were extrapolated to the mature forest area (estimated from 1:250,000 broad ecosystem mapping from B.C. Environment) in the range within British Columbia where Williamson's sapsuckers are relatively common.

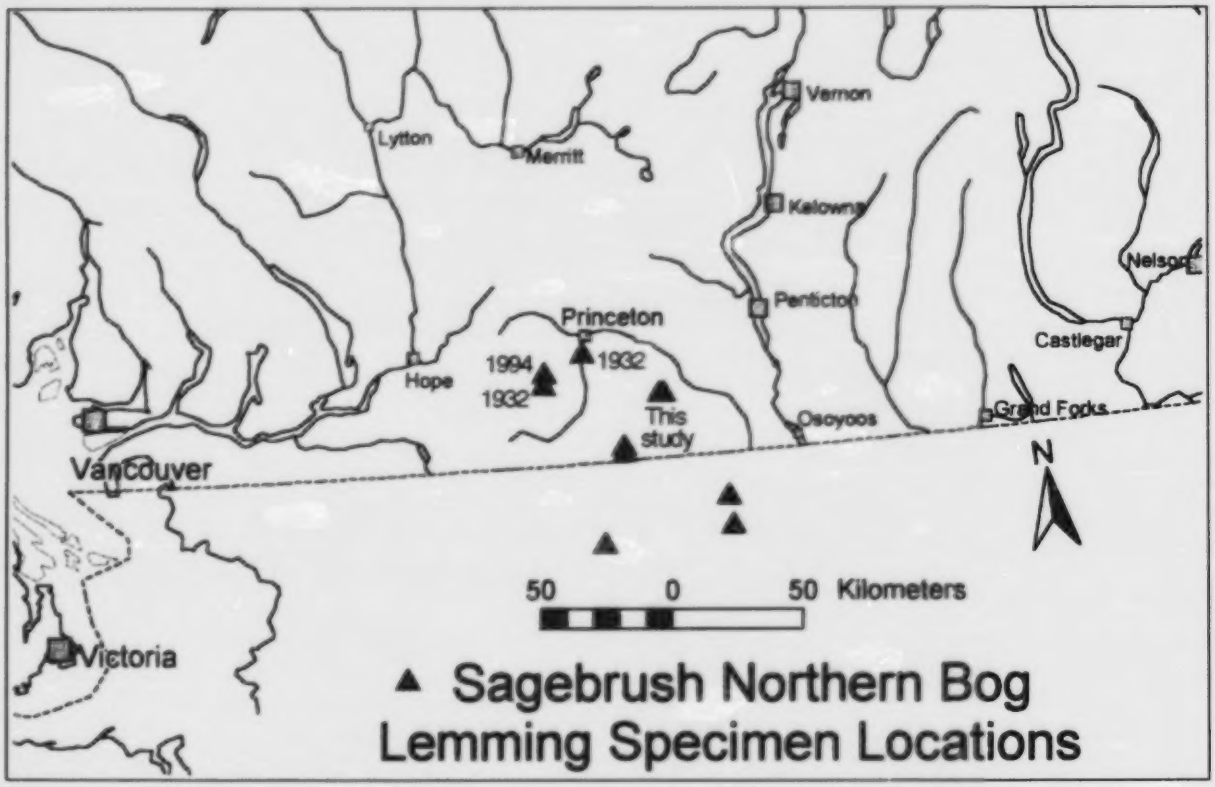

Figure 1. Sagebrush northern bog lemming (*Synaptomys borealis artemisiae*) specimen records. Canadian records from Anderson (1932), this study, and 1 specimen caught in 1994 by the senior author. U.S. records from Wilson et al. (1980) and from Washington State University specimen #81-398.

Mountain Beaver

Within the known range of mountain beavers on the east side of the Cascade Mountains, both reconnaissance and intensive surveys were undertaken in areas proposed for timber harvest in 1996–1998 FDPs. Reconnaissance surveys were done by first identifying the wet forested sites (e.g., stream channels, seepage channels, alluvial fans, and seepage slopes) that could be seen on aerial photos or suspected based on 1:20,000 contour maps. Intensive surveys attempting complete census of nest sites were undertaken where mountain beaver nest sites had been found on reconnaissance surveys and where mountain beavers were known or suspected to be found within finalized timber harvest boundaries. Occupied mountain beaver nest sites were recognized by "hay piles" of fresh or wilting green vegetation piled outside burrow entrances that were usually associated with radiating tunnel and runway systems.

In 1997, a complete census of nest sites was also attempted within (1) a 1,225-ha contiguous portion of the Champion Creek drainage where first-pass logging was largely complete, and (2) smaller areas of 40, 31.5, and 23 ha. This census was to determine if mountain beavers occupied clearcuts and if that occupation depended on the type of harvesting methods or silviculture treatments.

RESULTS

Sagebrush Northern Bog Lemming

In 1996, at Paul Creek, 5 bog lemmings (1 adult female, 4 juveniles) were caught. In 1997, at Easygoing Creek, 3 bog lemmings (3 adult males) were caught. No bog lemmings were trapped by either snap trapping or live trapping at Paul Creek or the headwaters of Smith Creek in 1997. All specimens were prepared as flat skins and skeletons and donated to the Royal British Columbia Museum. Habitats where specimens were caught included bog birch fens, hummocky moss fens, and seepage fens (receiving sites) in subalpine creek bottoms. All specimens were caught above 1,600 m elevation. Total lengths (TL) of adults ranged from 120 to 125 mm.

Williamson's Sapsucker

Despite wide-ranging surveys from 1996 to 1998, Williamson's sapsuckers were found only in a limited geographical area east of the Okanagan Valley, west of Midway, and within 40 km of the U.S. border (Fig. 2). Twenty-seven breeding season records of Williamson's sapsucker were tallied from 1996 to 1998 (counting pairs at a nest in a single season as 1 record, or individuals located where no nests were found as 1 record). Sixteen of these were from the Kettle dry mild Interior Douglas-fir (IDF) subzone (IDFdm1); 10 from the Okanagan very dry hot IDF subzone (IDFxh1); and 1 from the Okanagan dry mild Montane Spruce (MS) subzone (MSdm1). Elevation of the records ranged from 1,000 to 1,463 m, with most of the records (74%) between 1,000 and 1,200 m.

In all, 20 Williamson's sapsucker nests were discovered from 1996 to 1998 in 16 individual nest trees. Overall, Williamson's sapsucker nests were most commonly (65%) found in very large (70–118 cm diameter at breast height [dbh], 23–33 m height [ht]) live western larch (*Larix occidentalis*) with broken or dead tops. Next most common (15%) were small and dead paper birch (*Betula papyrifera*) (26–33 cm dbh, 11–20 m ht). Four of 10 nest trees checked in successive years were reused for nesting.

Overall density of occupied Williamson's sapsucker nests appears to be low, even in "ideal" habitat that has many potentially suitable nesting trees (e.g., large dead-top western larch). In 1997, at Woodlot 411-1 at Johnstone Creek, 3 nests and 2 drumming males were found for an estimated 5 breeding territories in an area of 600 ha. In 1998, we found 4 nests in a 500-ha area of Woodlot 411-1. Density was estimated at 1 nest/120 ha in 1997, and confirmed at 1 nest/125 ha in 1998.

Based on our surveys and those of Gyug and Bennett (1995), the estimated area where they are common within their range is 57,390 ha (Fig. 2), of which only 24.8% is forest greater than 120 years old, and might be considered Williamson's sapsucker habitat. Given that their maximum observed (confirmed) nest density is only 1 per 125 ha in the best of habitats in this area, then we estimate there could be up to 114 nesting pairs in this part of their range in Canada.

Mountain Beaver

We found 634 occupied mountain beaver nests from 1996 to 1998. Most (81.4%) of the nests were found in the moist warm Engelmann Spruce–Subalpine Fir (ESSF) subzone (ESSFmw); followed by 13.1% in the dry cold Thompson ESSF subzone (ESSFdc2); 5.0% in the Thompson dry mild MS subzone (MSdm2); 0.32% in the Cascade dry cool IDF subzone (IDFdk2); and 0.16% in the ESSF very dry cold subzone (ESSFxc). Most nests (81%) were found between 1,200 and 1,600 m elevation (range 800–1,925 m). Suitable mountain beaver habitat was best described as deep, wet or damp soils near drainage features, usually small streams or seeps. Most (93%) of the 329 tunnel/runway systems for which complete data were collected were in subhygric to hygric sites. Only 4% were in soil moisture levels as dry as mesic, and none were found in drier sites. In 23% of the tunnel/runway systems, the occupied nest site was found in a slightly drier site series than the most of the tunnels/runways used for foraging, usually being higher up on a gully edge, on raised hummocks in wetter ground, or on the edge of seepage areas.

We discovered 197 occupied nest sites in the intensive searches in the Champion Creek drainage. Densities of occupied mountain beaver nests were estimated at 30/100 ha in the north side of the 1,225-ha and 6.6/100 ha in the drier

southern portion. Estimated densities of active mountain beaver burrows in the smaller areas of higher quality habitat were 34.9, 30.4, and 35.0 per 100 ha. Mountain beavers did occupy clearcuts (31 of 197 nests) but densities were highly dependent on the type of logging and on silviculture treatments. Streams or wet areas that had been highly impacted by machinery during logging or site preparation were almost completely unoccupied by mountain beavers. Where trees had been hand-felled and then helicopter yarded or skidded over deep snow, densities were 43/100 ha. Where trees had been hand-felled or cut by feller-buncher and had been cable yarded, densities were 14/100 ha. Where trees had been harvested by feller-buncher, ground skidded, but no-machine buffers had been placed on streams, densities were 7.3/100 ha. Where there were no stream buffer zones, and most streams had been ditched or highly altered during harvesting and site preparation, densities were 1.7/100 ha.

Because the exact eastern range distribution of mountain beavers in Canada was only poorly known before these surveys, we have mapped here all our survey data (634 nests) as well as museum specimen records (Fig. 3). We have also divided the Canadian range of mountain beavers into 2 parts. First is the core part of their range where they are relatively common and found in most areas of suitable habitat. Second is a fringe area where they are uncommon, and are often absent from apparently suitable habitat.

DISCUSSION

Sagebrush Northern Bog Lemming

The specimens of northern bog lemmings recently caught (including the adult male specimen the senior author caught in 1994; TL 117), are more similar in total length (117–125) to previous specimens of *Synaptomys borealis artemisiae* (113–122) or to the Rocky Mountain subspecies *S. borealis*

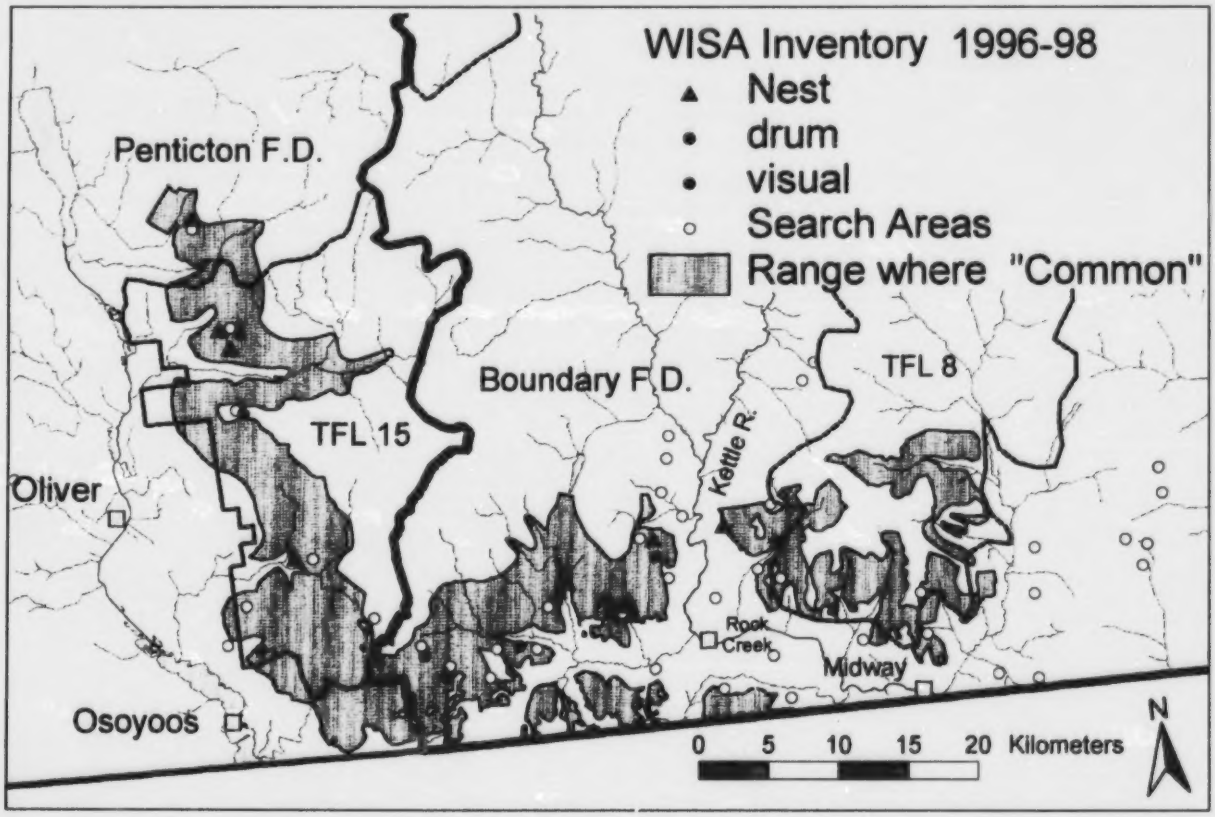

Figure 2. Williamson's sapsucker locations in the Southern Interior of British Columbia, 1996–1998, and the range where the species may be considered "common" reaching confirmed densities up to 1 nesting pair/125 ha in suitable habitat between 1,000 and 1,400 m in the Kettle dry mild Interior Douglas-fir (IDF) subzone (IDFdm1); Okanagan very dry hot IDF subzone (IDFxh1); or Kootenay moist cool Interior Cedar–Hemlock subzone (ICHmk1).

chapmani (114–132) than to the coastal subspecies *S. borealis wrangeli* (129–140), or the northern British Columbia subspecies, *S. borealis dalli* (131 average) (comparative data from Anderson 1932). However, there is no known zone of contact between *S. borealis artemisiae* and *S. borealis chapmani* so that *S. borealis artemisiae* is probably a valid subspecies.

Sagebrush northern bog lemmings have now been found in 8 localities including the 3 from northern Washington State (Fig. 1). Seven of these localities were characterized as bog birch or willow wetlands, or as wet meadows, which is more typical of the species throughout its range. The type specimens collected in sagebrush habitat near Stevenson Lake in 1927 (Anderson 1932) now are the anomaly in terms of habitat rather than the norm. The subspecific name "sagebrush" (*artemisiae*) is probably misleading and it appears that the subspecies habitat preferences are more similar to the other subspecies of northern bog lemmings rather than to any "semidesert" affinities. Perhaps the subspecies should be renamed "Cascade northern bog lemming" to more accurately describe its geographical distribution.

Williamson's Sapsucker

The preservation of large live western larch wildlife trees would appear to be very important to Williamson's sapsuckers in Canada. Any timber harvesting proposed in the range where they are common should target these large larch (especially those with broken tops) for wildlife tree patches. The estimate of 114 nesting pairs is probably an overestimate because large western larch have already been high graded from many of the mature forests within their range. An estimate of 100 nesting pairs in their entire range in Canada is probably more realistic. Based on the limited geographic range where they can be considered common, their low estimated numbers even within that range, and their apparent dependence on very large western larch in this area (this study, Cannings et al. 1987, Gyug and Bennett 1995), Williamson's sapsucker appears deserving of its Blue-listed status.

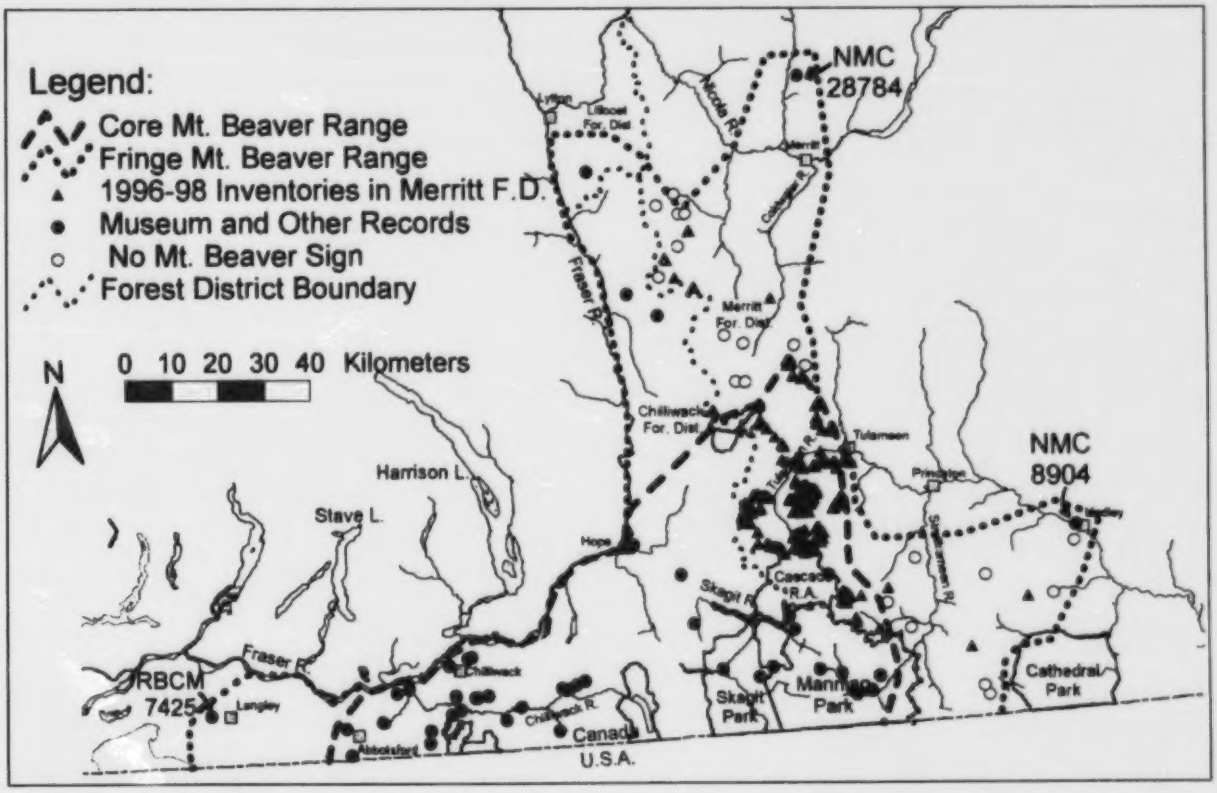

Figure 3. Eastern boundary of mountain beaver range within Canada based on 634 nest site locations obtained during 1996–1998 inventories on the east side of the Cascade Mountain divide. North American museum specimen records supplied by David Nagorsen (Royal B.C. Museum), and other observations supplied by naturalists or B.C. Environment staff. Limital museum specimens records are labelled RBCM (Royal B.C. Museum) or NMC (National Museum of Canada). See text for range definitions.

Mountain Beaver

Based on the evaluations made in Champion Creek in 1997, mountain beavers did appear susceptible to crushing of their nest sites and tunnel systems by heavy machinery. However, where machinery use was limited to deep snowpack, or where no machines were used, mountain beavers would survive after clearcutting, or reoccupy existing tunnel and burrow systems. Assessment of harvesting impacts though pre- and post-harvest censuses of mountain beavers in clearcuts began in 1998 on 2 small blocks, and will possibly continue in the coming years as more of the sites in which we found mountain beavers are logged. The recommended steps to protect mountain beavers in areas in the Merritt Forest District to be logged are (briefly): (1) delineate mountain beaver tunnel/runway system perimeters; (2) allow no machine use within these perimeters except on top of a deep snowpack; (3) fell trees by hand in the absence of a deep snowpack, and top-skid trees out; and (4) allow no mechanical site preparation within these perimeters because any machine use will collapse nests and tunnels.

ACKNOWLEDGEMENTS

This was a project of B.C. Environment Southern Interior Region with funding from Forest Renewal B.C. We thank J. R. Munro for field assistance throughout the project.

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Breeding Distribution of the Lewis's Woodpecker in the East Kootenay Trench in Relation to Fire History

John M. Cooper
Manning, Cooper and Associates
1278 Laurel Road, Sidney, BC, V8L 5K8, Canada
jcooper@islandnet.com

Cameron Gillies[1]
Manning, Cooper and Associates
1278 Laurel Road, Sidney, BC, V8L 5K8, Canada

ABSTRACT

The East Kootenay Trench of British Columbia was surveyed for nesting Lewis's woodpeckers (*Melanerpes lewis*) in June and July of 1997 and 1998. Forty-seven active nests were found in 1997, and 85 were found in 1998. Nest trees were mainly burned ponderosa pine and Douglas-fir snags. Nests occurred in 4 habitat types: burned coniferous forest, open mature ponderosa pine forest, grassland with isolated veteran ponderosa pines, and riparian cottonwood. Distribution of nests was clumped indicating that preferred habitat was not randomly distributed. In 1998, 59% of nests found were in burns that had suffered stand-destroying fires and 19% were in open, mature ponderosa pine forest with an obvious history of frequent fires. The remainder were found mainly in remnant ponderosa pine vets or riparian black cottonwood. Only a few hundred hectares of open mature ponderosa pine forest which is still suitable for breeding Lewis's woodpeckers exists in the East Kootenay Trench, but an estimated several thousand hectares of mature or older ponderosa pine stands could be enhanced for breeding habitat. An unknown percentage of these stands has likely become unsuitable for Lewis's woodpeckers because of in-filling of younger-aged trees. Currently, ponderosa pine or Douglas-fir forests that suffer stand-destroying fires provide the bulk of available breeding habitat.

Key words: black cottonwood, breeding habitat, Douglas-fir, East Kootenay Trench, fire, Lewis's woodpecker, *Melanerpes lewis*, *Pinus ponderosa*, ponderosa pine, *Pseudotsuga menziesii* var. *glauca*.

The Lewis's woodpecker (*Melanerpes lewis*) is an uncommon woodpecker of dry, open forests in the southern Interior of British Columbia (Campbell et al. 1990), where it reaches the northern limit of its range (American Ornithologists' Union 1983). A recent status report (Cooper et al. 1998) has suggested that populations are declining province-wide and, indeed, breeding populations have disappeared from the coast of British Columbia (Campbell et al. 1990). Consequently, the Lewis's woodpecker is on the provincial Blue List as a candidate species for designation as vulnerable (B.C. Ministry of Environment, Lands and Parks 1998).

This woodpecker requires open coniferous and/or deciduous forest with relatively large diameter dead or dying trees for breeding habitat (Sousa 1983). In British Columbia, most pairs build nests in large, decayed ponderosa pine (*Pinus ponderosa*) or black cottonwood (*Populus balsamifera*) (Campbell et al. 1990) and tend to breed in valley bottoms and foothills rather than on mountain slopes. In general, burned ponderosa pine forest is considered optimal habitat (Tobalske 1997).

The status of the Lewis's woodpecker in British Columbia is relatively well known in the Okanagan/Thompson region (e.g., Cannings et al. 1987, Campbell et al. 1990). Small numbers were thought to breed in the East Kootenay region, but that population's distribution and abundance was uncertain (Cooper et al. 1998). Surveys were designed mainly to determine breeding distribution, habitat use, and potential for habitat enhancement in the East Kootenay Trench ecosection, but this paper addresses the relationship of fire history to breeding distribution.

STUDY AREA

The study was conducted in the East Kootenay region of southeastern British Columbia. Surveys were restricted to the East Kootenay Trench (EKT) ecosection from the Canada/U.S. border north to Spillamacheen. The EKT is a relatively narrow ecosection that runs roughly north/south

[1] Present address: Tierra Consulting, Box 245 Windermere, BC V0B 2L0

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

from the border to Golden and includes the valley bottom and lower slopes of the Kootenay River and Columbia River valleys. The EKT also contains all of the Ponderosa Pine (PP) biogeoclimatic (BEC) zone and virtually all of the Interior Douglas-fir (IDF) BEC zone in the Southern Interior Mountains Ecoprovince (Demarchi 1995).

The PP zone occurs at the lowest elevations, is dry and very warm in summer, and has forest dominated by ponderosa pine. Stands are often open and stand structure is strongly affected by fire. Open grasslands occur frequently. Douglas-fir (*Pseudotsuga menziesii*) occurs on moister sites, trembling aspen (*Populus tremuloides*) occurs in riparian areas, and black cottonwood occurs on floodplains. The IDF zone occurs from the valley up to about 1,400 m in the EKT, is dry and warm in summer, and has forest dominated by Douglas-fir. At lower elevations and on dry sites ponderosa pine occurs frequently. Lodgepole pine (*Pinus contorta*) and western larch (*Larix laricina*) occur frequently in stands with frequent crown fires. Grasslands also occur frequently (Meidinger and Pojar 1990).

METHODS

The EKT was surveyed for nesting pairs in June and July of 1997 and 1998. We surveyed areas where we knew breeding occurred (from existing data or communication with knowledgeable persons), or in areas that appeared to have suitable habitat. About 125 person-days of surveys were conducted over the 2 years.

We believe that most areas in the EKT with suitable habitat were surveyed at least once, and that in 1998, we had surveyed all areas with concentrations of breeding pairs, unlike in 1997. Therefore, rather than combining results from 1997 and 1998, results from 1998 are considered to more accurately reflect real breeding distribution.

Nests were found by (1) either searching suitable habitat for adult birds, then, once having located them, watching until they returned to the nest cavity; or (2) by walking through suitable habitat and tapping on snags with cavities to flush birds from their nest. Lewis's woodpeckers are relatively wary compared with most other woodpeckers in British Columbia, and most birds on nests left their cavities as we approached the nest tree. Birds that were feeding young would were cautious about returning to their nests if we were nearby. In some cases, nests were easily located, in others, repeat visits were required to locate the nest tree.

Data on nest trees and associated habitat were collected as required on the provincial Ecosystem Field Form and Wildlife Tree Assessment Form. Data on fire history and extent of ponderosa pine forest in the East Kootenay Trench were obtained from files at the British Columbia Ministry of Forests, Research Branch.

RESULTS

Distribution of Breeding Areas

Breeding localities were distributed within the EKT from near Invermere, south to Newgate near the Canada/U.S. border. Forty-seven nests were found in 1997 and 85 were found in 1998; 60% of the 1997 nests were reoccupied in 1998. Most nests in 1998 were concentrated in 4 areas: Finlay Creek Burn (36%), Newgate (21%), Wycliffe (13%), and Dutch Creek Burn (8%). The remaining nests were found from Ta Ta Creek and Wasa north to Skookumchuck (9% in scattered locations), at the Fairmont Golf Course (7%), and east of Lake Koocanusa (5%) (Fig. 1). Most nests were clumped in a few general locations within the EKT and large tracts of the EKT appeared to have no breeding pairs.

Nests were found from valley-bottom elevation of 750 up to 1,110 m; 79% of nests were found between 750 and 1,000 m, and 21% were found between 1,000 and 1,110 m. All nests above 1,000 m were in burns. Fourteen of the 18 nests found above 1,000 m were in the Finlay Creek Burn. In this burn, a few birds were observed at elevations above 1,200 m, but no nests were found. We concluded that those birds may have been nonbreeders, or failed breeders, that may have moved to slightly higher elevations to forage. No birds were found above 1,200 m, even in large burns that appeared to provide usable habitat. For example, the Ram Creek Burn, which is about 15 km southeast of the Finlay Creek Burn and burned in the same year, destroyed most stands and created many square kilometres of open habitat with numerous snags. This burn is above 1,400 m and no nesting birds were found.

Breeding Habitat

General Breeding Habitat Classes

Nests were found only in the PP (38%) and IDF (62%) zones. Nest trees occurred in 3 broad habitat classes: ponderosa pine (37%), Douglas-fir/ponderosa pine (53%), and riparian (10%). Nesting habitat was also classified in a more descriptive manner: recent (<30 years) burns that had a stand-destroying fire (59%), open ponderosa pine forest (19%), riparian (13%), and grassland with scattered trees (8%) (Table 1).

Near Wycliffe, Wasa, and Ta Ta Creek almost all habitat depends on large dead or decayed ponderosa pine nesting trees, which are rare habitat features. At Finlay Creek, Dutch Creek, and east of Lake Koocanusa, almost all habitat is associated with stand-destroying burns that have left numerous snags for nesting. The only location where nesting occurs in open, repeatedly burned but live ponderosa pine forest is near Newgate.

Nest Trees

Most nests were found in ponderosa pine (40%) or Douglas-fir (35%) trees or snags, but also in trembling aspen (7%),

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999.

paper birch (*Betula papyrifera*) (5%), black cottonwood (4%), power poles (4%), or western larch (4%). Nest trees tended to be relatively large and decayed. Of 34 ponderosa pine nest trees, 21 were dead (12 were burned) and 13 were live. Of 30 Douglas-fir nest trees, 29 were burned and dead, and 1 was live but deformed (by lightning). We found noticeable differences in mean tree height, diameter at breast height (dbh), cavity height, presence of bark, and decay class between ponderosa pine and Douglas-fir nest trees. Ponderosa pine nest trees were taller (15.5 vs. 8.5 m), had larger dbh (59 vs. 48 cm), and were less decayed (3.5 wildlife tree decay class vs. 6.2) compared with Douglas-fir nest trees. Much of this difference is explained by all but one Douglas-fir nest tree being burned, dead, and decayed whereas many ponderosa pine nest trees were live or in very large dead vets.

Extent of Suitable Habitat

The EKT runs more or less north/south through 3 timber supply areas (TSAs). Estimates of forest area with ponderosa pine as the leading tree species in the 3 TSAs are Golden, 0 ha; Invermere, 5,448 ha; and Cranbrook, 21,185 ha. These

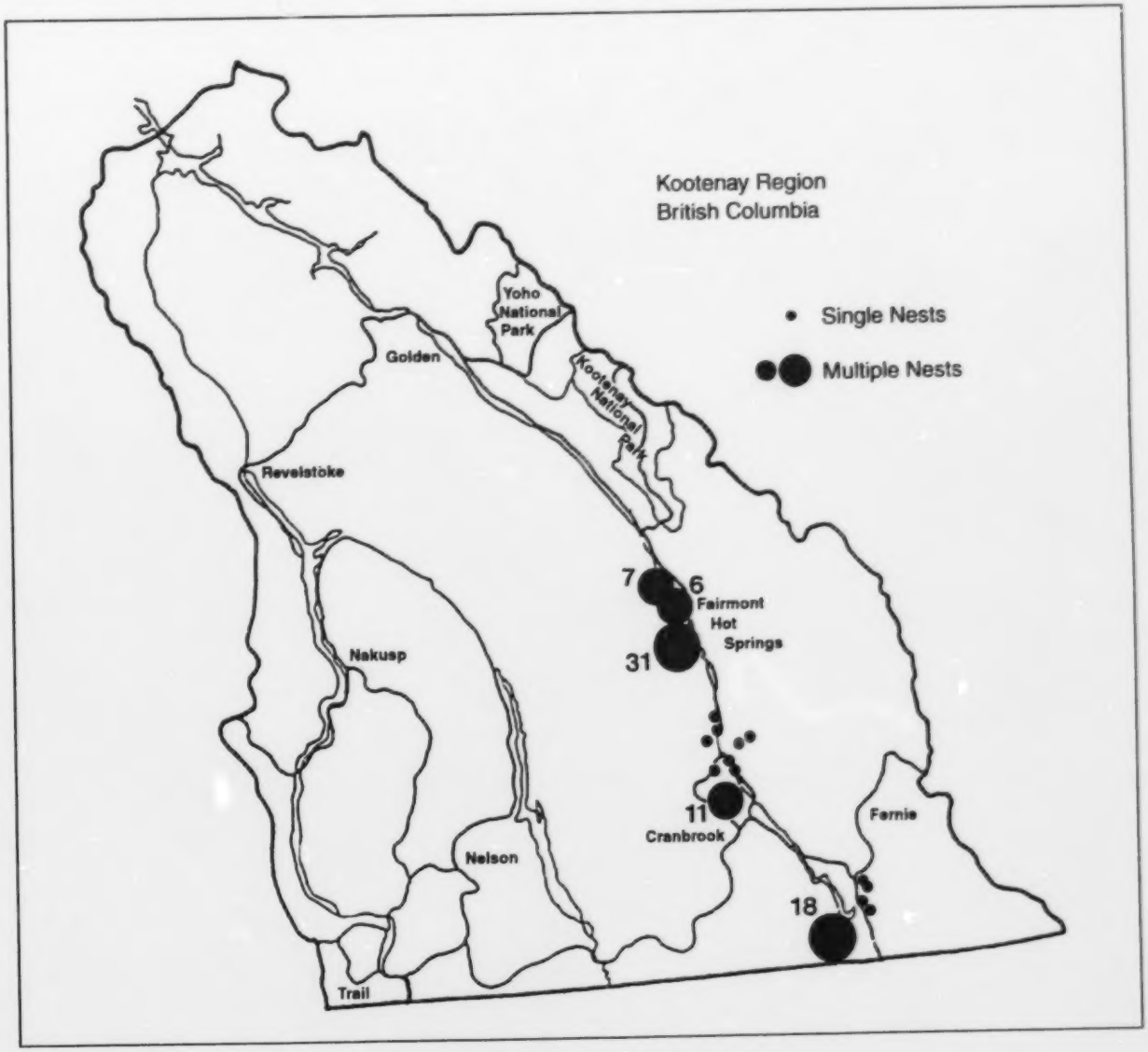

Figure 1. Locations of 85 Lewis's woodpecker nests in the East Kootenay Trench, 1998.

Table 1. General habitats used by nesting Lewis's woodpeckers, 1998.

	All nest trees	Pond. pine	Douglas -fir	Western larch	Black cottonwood	Trembling aspen	Paper birch	Utility pole
Biogeoclimatic zones								
Ponderosa pine	32	18	8	0	1	4	0	1
Interior Douglas-fir	51	16	22	3	2	2	4	2
Broad habitat class								
Black cottonwood riparian	8	0	0	0	3	1	4	0
Ponderosa pine	31	18	8	0	0	4	0	1
Douglas-fir/Ponderosa pine	44	16	22	3	0	1	0	2
General habitat description								
Riparian	11	3	0	0	3	1	4	0
Grassland[a]	7	5	0	0	0	1	0	1
Open forest[b]	16	8	3	0	0	4	0	1
Recent burn[c]	49	18	27	3	0	0	0	1

[a] Open grassland with very scattered trees.
[b] Open stand with fire history.
[c] Stand-destroying burn <30 years old.

estimates are for Crown land only, and do not include stands on private land. Of the stands on Crown land, 17% (3,611 ha) in the Cranbrook TSA and 21% (1,143 ha) in the Invermere TSA are mature or older (>120 years) (B.C. Ministry of Forests 1998).

Considerable but unknown numbers of ponderosa pine occur on private lands in the EKT. Almost all of the largest (>80 cm dbh) snags used for nesting grew on private lands. Most of these trees were at the edge of open grasslands or farmlands.

Fire History in the EKT and Lewis's Woodpecker Occurrence

Between 1969 and the early 1990s, 44 fires >20 ha in size occurred in the EKT between the border with the United States and Brisco. Most fires were caused by human activity (31) or lightning (11) (B.C. Ministry of Forests unpubl. data). Many of these burns did not appear to have suitable habitat for Lewis's woodpeckers, and some were not surveyed because of access problems. Nine of the burns have habitat suitable for breeding but only 4 contained breeding pairs. One additional area that burned before 1969 (Wolf Creek) contained 1 breeding pair (Table 2).

DISCUSSION

Fire strongly influences the ecology of the PP and IDF zones. Stand-maintaining surface fires were common, occurring on average every 5–15 years in the PP zone; and on average every 10–20 years in the IDF. Crown fires are rare in the PP zone, but occur in the IDF on average every 150–250 years (Daigle 1996). Surface fires are relatively low intensity fires that burn surface debris, kill shrubs and seedlings, thin out

Table 2. Burned areas that appeared to contain suitable breeding habitat for Lewis's woodpeckers (LEWO), year and cause of fire, and numbers of nests found.

Burn location	Year of fire	Cause	Maximum no: LEWO nests found 1997 or 1998
Dutch Creek	1971 (1985)	Human	9
Finlay Creek	1985	Human	31
Lake Enid	1994	Unknown	0
Ram Creek	1985	Lightning	0
Wolf Creek	1957	Human	1
NW Marysville	1979	Human	0
St. Mary's Reserve	1963	Human	0
Wild Horse River	1986	Human	0
E Baynes Lake	1982	Human	2
Newgate	1970	Lightning	18

younger stands, burn off lower branches of larger trees, yet rarely kill larger trees. The result includes higher crowns, branchless trunks, more widely spaced and larger trees, open understories, and open space under the canopy, all features that seem to be preferred by nesting Lewis's woodpeckers.

Fire history explains much of the distribution of nesting Lewis's woodpeckers. All of the breeding pairs found near Dutch Creek (7 pairs) and Finlay Creek (31 pairs) were in areas that had suffered stand-destroying fire. In the Dutch Creek Burn, nesting birds used mainly decadent snags left standing after the 1971 burn. This burn has long been known to be a "hot spot" for nesting Lewis's woodpeckers (Cooper et al. 1998). However, snags used for nesting are in soft condition and will fall soon. In addition, regenerating forest will fill in the open spaces that Lewis's woodpecker requires. Both of these inevitabilities will make the Dutch Creek Burn unsuitable habitat for breeding. At Finlay Creek, all 31 nests were found in the 1985 burn, and 25 of those nests were concentrated within a 600-ha section.

At Newgate, breeding pairs (18 pairs) occurred in 1 of 2 types of habitats. Some nests were in burned forest (1970) where a few snags remained standing amidst wide open spaces. These burned snags are fragile; one of the 1997 nesting sites and several potential nest sites had fallen or were knocked over between the 1997 and 1998 field seasons. The rest of the nest sites were in live ponderosa pines in open stands or clumps that had obvious fire history (repeated quick fires that destroyed undergrowth but did not kill the mature trees). These stands looked to have had many small (<20 ha so did not appear in the B.C. Ministry of Forests database), low intensity fires (stand structure consistent with that resulting from repeated fires every 5–20 years).

Extent of Suitable Habitat

Open ponderosa pine stands are extremely scarce in the East Kootenay Trench. They are created through repeated low-intensity fires that occur every 5–20 years. For example, a site at Lake Koocanusa showed a mean fire interval of 6.8 years, and a site near Canal Flats showed a mean fire interval of 20.6 years (Daigle 1996). Probably the best and most perpetual, Lewis's woodpecker nesting habitat in the study area was in the Strauss Wildlife Reserve near Newgate, where repeated fires have created the conditions previously described. We estimate that of the about 4,700 ha of mature or older ponderosa pine in the EKT, only a few hundred hectares have habitat attributes suitable for breeding Lewis's woodpeckers. The remainder are thought to be unsuitable mainly because stem densities and canopy closure are too high.

Burns with stand-destroying fires supply considerable suitable breeding habitat, as shown by concentrations of nests at Dutch Creek Burn, Finlay Creek Burn, and parts of Newgate. But these habitats are temporary. Several studies have shown that burns are typically used from a few years after the fire to about 40 years post-fire (Bock 1970, Tobalske 1997, Linder and Anderson 1998). In the EKT, most burned areas are providing temporary breeding habitat because nest trees are dead and will fall soon. Assuming that fires will be suppressed, regenerating forests will choke the open spaces. Two of these areas last had fires in 1970 and 1971; nesting trees in both areas are fragile.

In the Okanagan Valley, 44% of nests were found in black cottonwoods (Cannings et al. 1987), whereas we found only about 4% in black cottonwoods in the EKT. In the Okanagan Valley, that result was thought to be partially influenced by observer bias towards searching of riparian areas for nests. In the East Kootenays, conversely, observer bias favoured coniferous forest, as riparian areas were largely inaccessible. However, several apparently suitable stands of black cottonwoods (e.g., St. Mary River, Elk River, Bummers Flats, Bull River, Baynes Lake) were surveyed and nesting Lewis's woodpeckers were found in only 1 stand. The limited data indicate that riparian forest is used less for nesting in the EKT than in the Okanagan Valley.

We found no Lewis's woodpeckers in 3 areas in the EKT that appeared suitable. The most puzzling of these was the lack of nesting pairs found on the east side of Lake Koocanusa. This area is directly across the lake from Newgate, where many pairs nested. Although unused nest holes were found in several areas with seemingly usable habitat, only 1 nest was found—in a small trembling aspen. A burn along the lower foothills of the Wild Horse River, near Fort Steele appeared to offer ideal habitat—hundreds of dead snags spread out over a large low elevation open area. We anticipated finding breeding pairs there but none was found in 1997 or 1998. An area near Wasa contained 2 apparent nest cavities but no birds were seen in 1997 or 1998.

MANAGEMENT IMPLICATIONS

Most nests were found on Crown land, including the areas with relatively high concentrations of nesting pairs. Management of these areas could, therefore, provide for Lewis's woodpecker breeding habitat. The most desirable habitat type is probably open and mature ponderosa pine forest. If relatively frequent fire regimes are allowed, this type of habitat may be maintained at various localities in the PP and IDF zones.

Breeding habitat could be enhanced through mechanical brushing of stands with mature stems but which are "choked" with regeneration, followed by selective logging of some stems, and, possibly, prescribed burning. Many hundreds of hectares could probably be enhanced for Lewis's woodpecker breeding habitat by these methods. Of an estimated 4,700 ha of mature or older ponderosa pine in the Invermere and Cranbrook TSAs, only a few hundred hectares could be currently classed as suitable habitat.

Some management potential for habitat on private land also exists. Many of the largest remaining ponderosa pine snags in the EKT occur on private lands. Most landowners were pleased that a rare woodpecker was living on their property; therefore public education may be used to encourage landowners to retain important habitat features such as large, decayed ponderosa pines. While these individual snags may stand for some time, they are exceedingly rare in the area, and are subject to cutting for firewood, as possible danger trees (for those trees near buildings), or for "aesthetic" reasons at the discretion of the landowner. Because most large ponderosa pine have been removed, once the trees with active cavities fall, there will be no live, large ponderosa pine trees to replace them, and pairs must either use alternate tree species or age classed, or not nest in the area.

Improved management following stand-destroying fires could help conserve breeding habitat for Lewis's woodpeckers. Standing burned trees are frequently harvested for timber or firewood, and areas are replanted. In the Finlay Creek and Dutch Creek burns, both of which were harvested post-burn, nesting Lewis's woodpeckers are often found only in the few remaining large diameter snags. Retention of more snags would increase the availability of potential nest trees. This could be best accomplished by leaving more snags during post-burn harvesting, and blocking road access to discourage firewood cutting.

If fire frequency remains low, then the relatively small amount of open and mature ponderosa pine forest in the EKT will inevitably decline and possibly disappear. In the long term, reduced fire decreases the health of fire-dependent ecosystems by making areas more susceptible to stand-destroying fires, and by changing diversity of plants and animals that occur there. Ultimately, Lewis's woodpecker may become even more dependent on stand-destroying fires to provide breeding habitat in the EKT.

ACKNOWLEDGEMENTS

The authors thank the Habitat Conservation Trust Fund for funding this project. Special thanks to M. Chutter (B.C. Ministry of Environment, Lands and Parks) for his encouragement and administrative support. M. Nyhof cheerfully worked long but happy days looking for nests, recording data, and solving problems. D. Cooper and J. Cooper found some Lewis's woodpeckers in unusual places. Many thanks to dozens of local naturalists and residents, and government personnel who took an interest in this project. B. Warkentin (B.C. Ministry of Environment, Lands and Parks) provided a great overview of the study area by driving us through most of the potential habitat in the East Kootenays, showing us the best access to remote sites, and saving us much time. Also thanks to B. Warkentin for his historical perspective on habitat changes in the East Kootenay.

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Threats Posed by Rock-Climbers to Birds Nesting on Cliffs in the South Okanagan

Howard Richardson
Biology Department, Okanagan University College
583 Duncan Avenue West, Penticton, BC, V2A 8E1, Canada
howie@vip.net

ABSTRACT

An explosive growth in the sport of rock-climbing has brought previously isolated, cliff-dwelling species into close contact with humans. Skaha Bluffs in the South Okanagan Valley, B.C. is now a major rock-climbing destination and home to at least 13 Red- and Blue-listed species. Of these, by far the most visible is the white-throated swift (*Aeronautes saxitalis*). Peak use of this area by both swifts and climbers is April to September. Most of the swifts' nests are on cliffs little used by climbers, who also prefer to climb on open faces rather than the cracks frequented by swifts. There was no detectable difference in the proportion of successful nests on cliffs used by climbers compared to unused cliffs. Nor was there any decrease over the course of the study in the total number of nesting swifts in Skaha Bluffs, or a move from cliffs popular with climbers to unvisited ones. Canyon wrens (*Catherpes mexicana*) seem equally unaffected by the surrounding clamour. The swift population is larger than previously determined, scattered over a large number of cliffs in the Okanagan. About 10% of the valley's nesting swifts are found in Skaha Bluffs. The same happy state of affairs may well not exist for other threatened species in the area. Climbers need to be involved in, and more aware of, potential problems and their resolution if climbing is to be a perennial part of the local, recreational scene.

Key words: *Aeronautes saxitalis*, canyon wren, *Catherpes mexicana*, Okanagan, rock-climbing, white-throated swift.

The first rock-climbs in British Columbia were done in the early 1950s, but for most of the next 30 years this was a sport practised by few and with little ecological impact. In the mid-1980s, 2 innovations impelled rock-climbing from a backwater activity into the mainstream. The first was the indoor climbing gym, where people could climb all year, and which provided an easy avenue of entry into the sport. The second innovation was the battery-powered hammer-drill. This allowed belays, or safety anchors, to be placed in blank walls in a matter of seconds. Hitherto, climbs had followed cracks into which protective devices could be placed, or had required a lot of time and effort to hand-drill holes for anchors to safeguard the blank sections. The 2 factors have worked synergistically, with ever more climbers looking for places to climb and using cliffs previously thought to be dangerously unprotectable.

This burgeoning web of climbing has engendered numerous repercussions on the grounds of aesthetics, access, and liability (e.g., McCarty 1993). Conservation issues have been slower to develop, perhaps because rock walls were traditionally seen as sterile, inaccessible, and ecologically uninteresting (but see Nuzzo 1995, Kelly and Larson 1997, Camp and Knight 1998). In Britain, where there is a high density of both climbers and naturalists, the avenues for dealing with conservation problems are well established. Climbers and conservation groups consult over possible cliff closures and the results are published in climbing magazines (Anon 1998). In North America such problems are dealt with on a more ad hoc basis.

The spread of rock-climbing and its potential impacts are nowhere more problematic than in the South Okanagan. In the past 10 years, Skaha Bluffs, just outside Penticton, has become the second most-popular climbing area in British Columbia. This within an area of Canada that is ecologically unique and already beleaguered by immigration, development, agriculture, and other recreational activities. Cliff faces, which for the most part had been undisturbed since they were uncovered at the end of the last ice age, are now visited on a frequent and regular basis. Eighteen of the South Okanagan's Red- and Blue-listed species are known to need cliffs or talus as an essential component of their habitat. Thirteen of these have been found in the climbing area.

Climbers, government agencies, and other concerned parties have made numerous attempts to broker agreements to minimize the impacts of climbers at Skaha Bluffs. Such

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp

attempts usually founder on lack of information about the species of concern. The aims of this study were: a) to determine the basic activity patterns within Skaha Bluffs of both climbers and white-throated swifts (*Aeronautes saxitalis*); b) to assess the impacts of climbers on the swifts' breeding success; and c) to evaluate the importance of Skaha Bluffs to the swifts on a valley-wide scale. The white-throated swift is known to breed at Skaha Bluffs and, with only 200 breeding pairs in Canada, is Blue-listed (Summers 1995). It and the canyon wren (*Catherpes mexicana*) are the most visible and probably the most common of the listed bird species found in the area.

STUDY AREA

Skaha Bluffs consists of about 45 separate rock outcrops scattered through 100 ha of ridges and canyons on the eastern outskirts of Penticton. Another 90 cliffs throughout the Okanagan, from Kelowna to the United States border, were visited to confirm the overall distribution and numbers of swifts in the area.

METHODS

The data were collected between late March and early November of 1995–98. More than 100 visits, lasting 4–16 hr, were made in each of the 4 years.

Activities of Swifts

Gaining access to swifts' nests is notoriously difficult. I managed to get to, and look into, 4 nest sites, and did so in each of the 4 years of the study. Visits were made about every 3 days; within this observation schedule, laying and hatching dates were more narrowly defined by assuming a 2-day interval between laying of successive eggs and synchronous hatching (Lack and Lack 1951, Baldwin and Zaczkowski 1963). The noisy and highly-visible copulation flights were assumed to be coincidental with laying and so were also used to determine the temporal distribution of clutch initiations.

In 1998 the activities of swifts were monitored by weekly visits to 79 nests on 14 cliffs. Approximately half of these nests were those in the Skaha Bluffs. The rest were on 3 cliffs near Skaha and Vaseux lakes chosen simply for their accessibility (UTM [universal transverse mercator] references: E 3123, N 54701; E 3139, N 54766; E 3163, N 54637). This provided a mix of cliffs used by climbers moderately, infrequently, and not at all. None of the cliffs most heavily used by climbers had swifts nesting on them.

When the loud begging calls of the young are no longer audible they have either fledged, been abandoned, or have died. Young were assumed to have fledged if all of the following conditions applied: a) adults visited the nest in all 3 of the weeks prior to the presumed fledging; b) the adults visited more than once an hour in those 3 weeks; and c) the young begged strongly at each visit.

Activities of Climbers

Details of the date of first ascent, location, grade, and general description of climbs came from the appropriate guidebooks to the area (Knight 1996, McLane 1995, Richardson 1997). The number of climbers visiting Skaha Bluffs was determined by counting cars in the car park and their occupants as they arrived.

Throughout 1996 I kept a tally of all the climbs where I saw people attempting an ascent. Because it was not possible to visit all the cliffs with equal frequency, popularity of a climb was expressed as number of ascents per observation-visit. Some cliffs are larger and have many more routes than others. Cliff popularity was expressed as number of ascents per total number of climbs on the cliff per observation-visit. Hence, a large cliff with 35 climbs would not be registered as being as popular as a cliff with few climbs, if they received equal numbers of ascents.

RESULTS

Activities of Swifts

Although white-throated swifts arrive in the Okanagan in late March, they do not start breeding until early May. A total of 109 copulations were recorded between the first week in May and the first week in July, with the peak in the second week of June (Fig. 1). Of the 79 regularly visited nests, only 59 were successful. The young fledged between the last week in July and the first week in September, with most fledging in the second week of August (Fig. 1). Most birds left the valley by mid-September. The duration of incubation and brooding, from direct observation of the 4 accessible nests, was 23

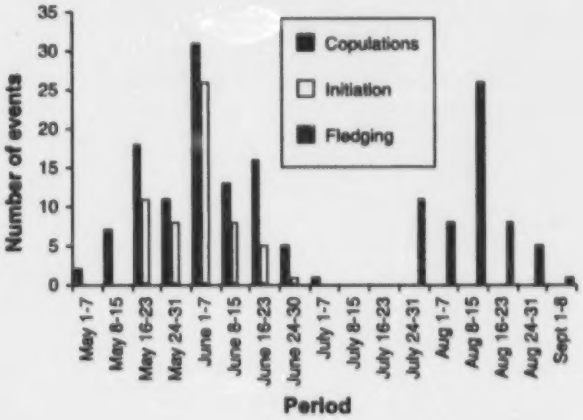

Figure 1. Breeding chronology of white-throated swifts at Skaha Bluffs.

days and 42 days, respectively. These durations are consistent with other swift species of similar size (Cramp 1985, Marin 1997). This suggests the 59 successful nests were initiated between early May and mid-June (Fig. 1), a period coinciding with that of copulation activity.

My observations in the summer of 1998, combined with previous surveys (Summers 1995), suggest a valley-wide white-throated swift population of around 750 birds, of which 68 (9.1%) were at Skaha Bluffs. In all, swifts were seen at 79 of the 135 cliffs visited.

Activities of Climbers

Both the number of cliffs in British Columbia used by climbers and the number of climbs on them have grown exponentially since the 1950s. There were recorded climbs on only 9 cliffs in the 1950s, but that number had reached 190 by the mid-1990s. Of particular note is that much of the development since 1985 has been in the Okanagan (Fig. 2).

In 1996 approximately 4,800 cars, with an average of 2.5 occupants per car, used the climbers' car park, giving roughly 12,000 climber-days spent at Skaha Bluffs. A variety of casual observations, such as the number of parking passes purchased, suggest visitation is growing by about 10% per year, so there may have been as many as 15,000 visits in 1998. Although there could be a few visitors on a sunny day even in the middle of winter, the period of consistent use (April to September) and heavy use (April and May) coincides almost exactly with the swifts' breeding season. On the busiest day there were 120 cars in the parking lot!

One of the common misconceptions of non-climbers is that once a climb is described in the guidebook it will then be climbed repeatedly. But such is not the case. Climbers show distinct preferences, resulting in some climbs and cliffs being heavily used while others may rarely be visited. During the summer of 1996 I never saw climbers at 15 (33%) of the 45 cliffs on which, at that time, there were recorded climbs. Only 12 of the 45 (27%) had climbers on them for more than 1 of every 10 visits. By contrast there was at least 1 climber present on 88 of 100 visits to the 2 most popular cliffs. None of the 4 most-heavily used cliffs had swifts nesting on them. Of the 11 cliffs with swifts' nests, 3 are never climbed, 4 receive few visits, and 4 were ascended with moderate frequency. The most popular (no nests), moderately popular, and infrequently visited cliffs had levels of use of >0.1, 0.05–0.3, and <0.02 ascents per climb per visit, respectively.

What was the basis for climber preference? They liked climbs with fixed protection anchors rather than ones where they had to ascend cracks and place their own protection. A total of 585 climbs were listed in the guidebook, of which 275 (47%) had fixed protection, compared with 989 observed ascents, of which 574 (58%) were of fixed climbs ($\chi^2 = 48.4$, $P < 0.001$). They also preferred climbs graded between 5.8 and 5.10 (Fig. 3), and only 2 of the climbs with swift nests on or near them were in that grade range. There was a weak negative correlation between cliff popularity and distance from the car park (cliff popularity [ascents/climb/visit] = $-1.1^{-5} \times$ distance[m] + 0.04; $r^2 = 0.04$). Much of this correlation is due to the 4 most popular cliffs, all of which are heavily used for teaching and by novice climbers. If these 4 cliffs are ignored, it seems that more experienced climbers are interested in some other quality of a climb than simply how far they must walk to reach it ($r^2 = 0.0001$).

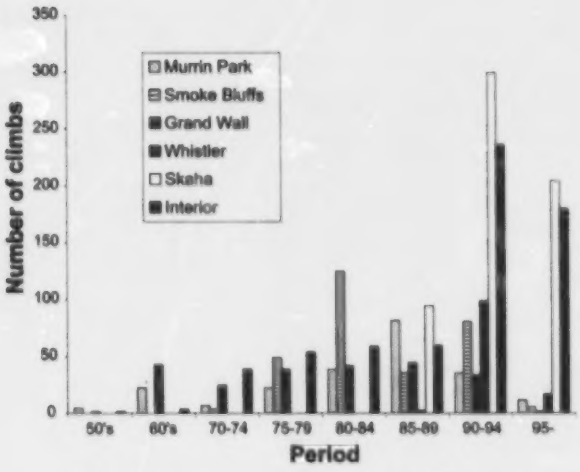

Figure 2. The number of new climbs recorded in 5-year periods in different areas of British Columbia.

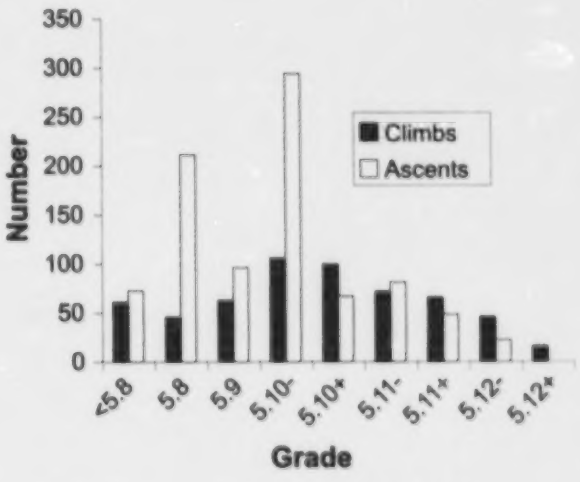

Figure 3. Frequency distribution of climbs of different grade and of actual ascents of climbs in those grades.

Do Climbers Affect Breeding Swifts?

The data allow 3 possible assessments of whether climbers had a negative impact on the breeding of white-throated swifts. The first was a comparison, for 1998, of the proportion of successful versus unsuccessful nests on 3 groups of cliffs: a) those outside the climbing area; b) cliffs in Skaha Bluffs that had climbs on them; and c) those that did not have climbs on them (Fig. 4). Although there were differences ($\chi^2 = 0.82$, $P = 0.38$) between the groups, the trend was contrary to expectations, with the cliffs in Skaha Bluffs with no climbs on them having the lowest proportion of successful nests. Secondly, there was no significant shift ($\chi^2 = 5.61$, $P = 0.47$) over the duration of the study of nesting activity from the most popular cliffs to those less frequently visited by climbers (Fig. 5). The third possible line of evidence for a minimal disturbance of these birds by climbers is that the overall number of swift nests in Skaha Bluffs has not declined over the 4 years of the study with, sequentially, 30, 32, 30, and 38 nests.

DISCUSSION

For conservationists there is good news and there is bad news. The good news is obviously that white-throated swifts do not seem to be impacted by this sudden incursion of climbers into their breeding habitat. This is most likely due to the separation of climbers onto the open faces and birds into the cracks, but also because swifts are known to be unconcerned by human activity close to their nests (Lack and Lack 1951). Also reassuring is the fact that climbers currently interact with less than 10% of the valley's swift population, and that the population is either growing or larger than previously estimated. Coincident observations of canyon wrens suggests they are similarly unperplexed by the flurry of human activity around them.

The bad news is that the number of climbs and climbers is still increasing. Ten years ago most climbers dismissed the Okanagan as having too little solid, climbable rock to be worth a visit. The survey undertaken in this study revealed numerous climber-friendly cliffs that have, as yet, escaped notice. With more and more climbers moving to the area specifically for the climbing, this situation will not continue.

The second item of bad news is that whatever can be said about the ability of swifts, wrens, and climbers to coexist cannot be extrapolated to any of the other species of concern. Snakes, bats, sheep, and raptorial birds present a completely different complex of interactions.

Climbers have historically viewed their activities as environmentally benign. This view has to be called into question given the escalating geographical scope of the sport recorded here, and the presence on a long weekend of up to 300 visitors in such a small area as Skaha Bluffs. But for this and other sites on Crown land, it is nearly impossible to implement either legislated or voluntary restrictions of human, recreational activity merely on the suspicion that it is detrimental. Such restrictions are only palatable under 2 conditions: that the potential damages are rationally and realistically presented; and that any restrictions can be seen to be part of an overall plan. Climbers have seen restrictions of their activity as unfair when 5 km down the valley a similar, and probably equally essential, piece of habitat is mutilated for a housing development.

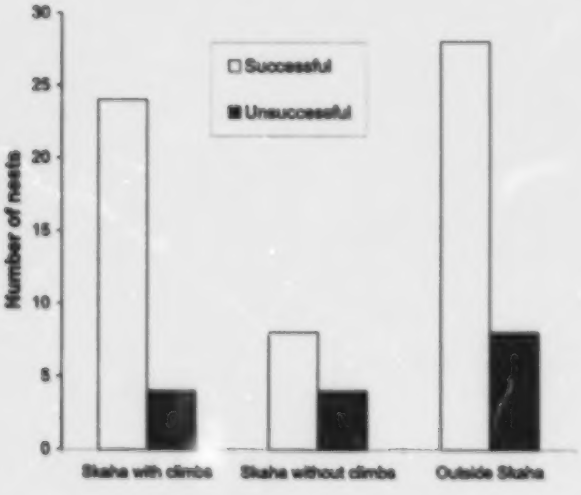

Figure 4. The number of successful and unsuccessful nests on cliffs in Skaha Bluffs with climbs, without climbs, and on cliffs outside the climbing area.

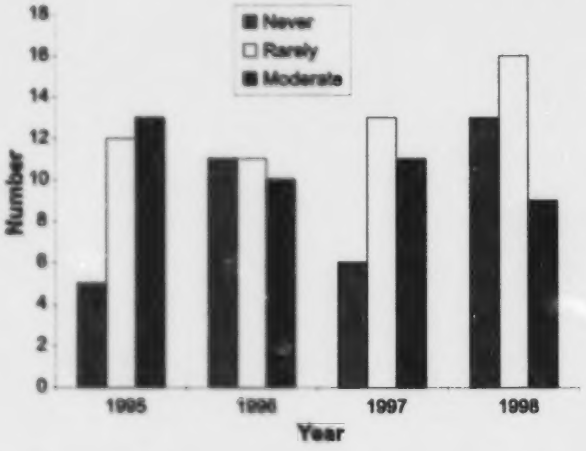

Figure 5. The number of swifts' nests per year on cliffs visited by climbers never, rarely, or with moderate frequency.

Acquiring information and formulating management plans requires time and money. The relevant government agencies do not have enough of either. If climbers want to continue pursuing their passion in the Okanagan they must become more aware, more involved, and more willing to accept restrictions. I have been a climber for 40 years; I would hate to have to stop now.

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Townsend's Big-Eared Bat (*Plecotus townsendii*[1]) in the East Kootenays

M.C. Firman
Consultant to Columbia Basin Fish & Wildlife Compensation Program
182 Fyffe Rd. SE, Calgary, AB, T2H 1C1, Canada
mitchell@petrodat.com

ABSTRACT

In Canada, Townsend's big-eared bat occurs only in British Columbia, where it is Blue-listed. Although found from Vancouver Island to the Rocky Mountains, this bat has a scattered distribution, usually associated with the dryer biogeoclimatic zones of the southern coast, southern interior and central interior. In Canada, little is known of this bat's biology. The Townsend's big-eared bat was rarely captured or detected by typical bat survey methods and was previously known from very few maternity colonies in the province. In 1996, a survey funded by the Columbia Basin Fish and Wildlife Compensation Program (CBFWCP) located a maternity colony of 100 Townsend's big-eared bats at the St. Eugene Mission (near Cranbrook). Previously unknown in the East Kootenays, this colony was also the largest ever found in the province and the only maternity colony ever found in the interior. Unfortunately the status of the abandoned St. Eugene Mission was uncertain, as it was to be developed as part of a golf resort. The CBFWCP funded further studies to determine the feasibility of incorporating the roost into the finished resort and to locate hibernacula and other roosts of this species in the East Kootenays. Resort developers approved the roost concept / design and it was built in the spring of 1997. The roost is remotely monitored and was successfully occupied in the summers of 1997 and 1998 although resort construction is not yet completed. Five additional maternity colonies (50 or less individuals) were also located in buildings as well as 4 abandoned maternity sites (including one natural roost). Five hibernacula have been located in mineshafts, with < 40 individuals occupying one site. Steps are being taken to preserve these additional maternity and hibernacula sites for the bats.

[1] Also known as *Corynorhinus townsendii*

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Ecological Characteristics of Inland Rain Forests

André Arsenault
British Columbia Ministry of Forests, Kamloops Forest Region
515 Columbia Street, Kamloops, BC, V2C 2T7, Canada
Andre.Arsenault@gems8.gov.bc.ca

Trevor Goward
Enlichened Consulting, Edgewood Blue
Box 131, Clearwater, BC, V0E 1N0, Canada

Key words: intermontane region, lichen, old growth, rain forest.

SUMMARY

British Columbia's Rain Forests

Within the northern hemisphere a major proportion of the world's rain forests at temperate latitudes occur along the west coast of North America. Fronting the Pacific Ocean, and centred in British Columbia at 43°N to 61°N (see also Alaback 1991), these coastal rain forests are characterized by a highly oceanic climate, with heavy precipitation, high overall humidity, moderate temperatures, and relatively infrequent thunderstorm activity (Agee 1993). Related to the above, they are also characterized by a low incidence of wildfire (Agee 1993, Arsenault 1995), a preponderance of old-growth forests, a biomass virtually unequalled in other terrestrial ecosystems (Waring and Franklin 1979), and a rich epiphytic nonvascular flora. Outside of British Columbia, the coastal rain forests narrow southward into Oregon, and northward into southeast Alaska.

The coastal rain forests of western North America have long provided an international flashpoint for environmental concern. By contrast, their inland counterparts—the rain forests of intermontane British Columbia—are still poorly known, even to researchers. Located between 51°N and 54°N along the windward slopes of the Columbia and Rocky mountains (Fig. 1), the inland rain forest phenomenon is unique to British Columbia. It is restricted to a region of anomalously humid climate, in which a plentiful snowmelt during late spring is followed by ample rainfall during the height of the growing season. More specifically, inland rain forests are confined to the wettest subzones of the Interior Cedar–Hemlock zone (i.e., the ICHwk and ICHvk); in no other region of the world has a similar integration of humidity and continentality been documented.

The Rain Forest Project

In 1995 we initiated a project funded by Forest Renewal British Columbia (FRBC) to compare and contrast inland rain forests with their coastal counterparts. This project had 2 components: 1) to study similarities and differences in disturbance ecology and forest dynamics; and 2) to discern differential patterns of lichen and bryophyte diversity. This project evolved from earlier work on pattern and process in the coastal temperate rain forests (Arsenault 1995, Arsenault and Bradfield 1995), and on the distribution of rare macrolichen species (Goward 1994, 1996; Goward et al. 1994).

We adopted a multiple-scale approach in order to gain a comprehensive understanding of rain forest dynamics and species distribution. The first author examined disturbance ecology and forest dynamics at 3 scales of ecosystem organization: 1) at a within-stand scale assessing gap-phase dynamics; 2) at a meso-scale using forest chronosequences to compare stands of different ages; and 3) at a landscape scale examining the distribution of major disturbances and plant communities. Our study of lichens was similarly organized: 1) at a within-stand scale examining microdistributional ecology (Arsenault and Goward 2000); 2) at a meso-scale focusing on differences in species distribution and abundance in forests of different ages; and 3) at broader geographical scales developing predictive tools for the recognition of

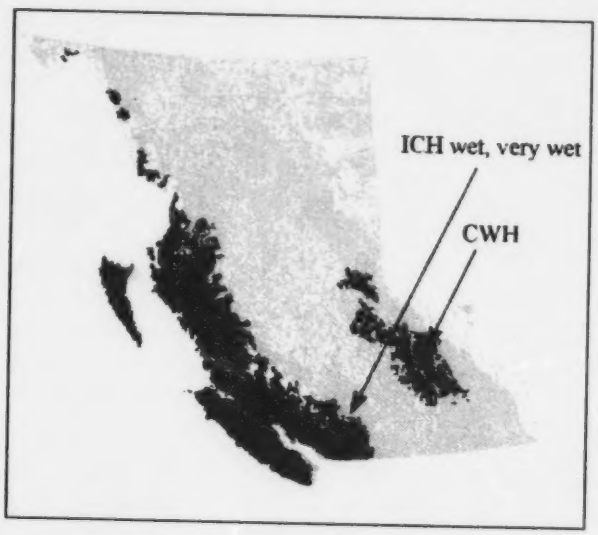

Figure 1. The occurrence of coastal and inland rain forests in British Columbia (CWH = Coastal Western Hemlock biogeoclimatic zone; ICH = Interior Cedar–Hemlock biogeoclimatic zone).

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lichen hot spots (Goward and Arsenault 2000). A similar approach was used for bryophytes in a collaborative study with colleagues from the University of Alberta.

Oceanic Species

By their high ambient humidity, inland rain forests favour colonization by numerous oceanic species, that is, species more often associated with maritime environments. For example, hanging moss (*Antitrichia curtipendula*) forms thick hanging mats on the boughs of western redcedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) trees. A number of oceanic vascular species also occur here, including deer fern (*Blechnum spicant*) and red huckleberry (*Vaccinium parviflorum*). Perhaps most fully represented, however, are various oceanic epiphytic lichen genera, including *Chaenotheca*, *Chaenothecopsis*, *Collema*, *Fuscopannaria*, *Lichinodium*, *Lobaria*, *Nephroma*, *Parmeliella*, *Polychidium*, *Pseudocyphellaria*, *Sphaerophorus*, and *Sticta*. Many species belonging to these genera are old-growth-dependent in this portion of their range, and are rare or infrequent in British Columbia as a whole.

Forest Dynamics

Dominated at maturity by western redcedar and western hemlock, inland rain forests closely resemble coastal temperate rain forests in the complexity of their architecture, and the lushness of their understories. They differ, however, in their respective disturbance regimes and regeneration dynamics. Although rain forests as a whole are characterized by a low frequency of catastrophic disturbance, inland rain forests are proportionately much more susceptible than coastal rain forests to fire, wind, insects, and avalanches.

Figure 2. Landscape mosaic in the Upper Adams River Valley.

Related to this, the age-class structure of inland rain forest landscapes is somewhat less weighted to old growth (Fig. 2 and Fig. 3), and even less to "antique" forests, that is, forests that have escaped catastrophic disturbance for a period longer than the age of the oldest trees within them (Goward 1994).

Management Implications

Perhaps our most significant finding to date is that rare species are not uniformly distributed in old-growth forests, but tend to associate with forests characterized by us as antique. Similarly, antique inland rain forests are themselves not distributed randomly across the landscape; rather they are situated in gullies, toe positions, and other moist to hygric sites that tend repeatedly to escape wildfire. This observation would seem to have important implications for the establishment of future old-growth management areas.

We suggest that the oldest old-growth rain forests of inland British Columbia are at risk. More specifically, their future contribution to biological diversity may be diminished as a result of several interacting factors, including: 1) their restricted historic distribution; 2) their association with highly productive sites; 3) their accessibility to resource extraction; and 4) their past and ongoing fragmentation as a result of timber harvesting, natural disturbance, and flooding for dam construction. The insights generated by our project have already assisted in the designation of protected areas; we hope they will continue to contribute to landscape unit planning and future forest practices.

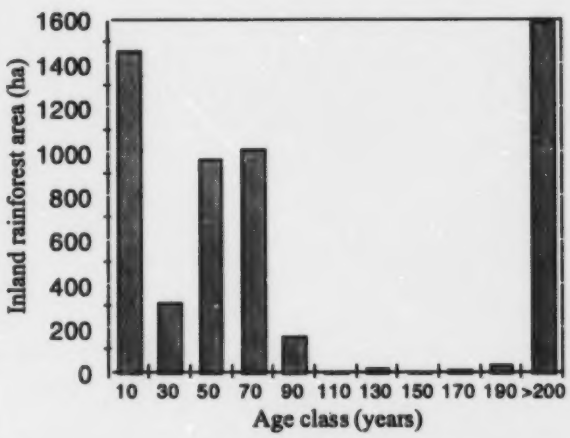

Figure 3. Stand age distribution for the ICHwk and ICHvk (Interior Cedar–Hemlock wet and very wet) subzones, Upper Adams River Valley. The first age class (0–20 yr) results from recent clearcut harvesting. Most of the stands between 40–120 years old originate from catastrophic fire that burned substantial portions of the area near Tumtum Lake.

ACKNOWLEDGEMENTS

We wish to extend our sincere gratitude to K. Johnston, P. Williston, and G. Yearsley for capable assistance in the field, to D. Lloyd for helpful discussions on this topic, and to S. Witt for assistance with the ortho-photograph and forest cover maps. Funding was provided by Forest Renewal British Columbia and by the British Columbia Ministry of Forests.

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Northern Goshawk Inventory of the Fort Nelson Forest District

S.N. Bennett

Mirkwood Ecological Consultants Ltd.
501 10th Ave., Castlegar, BC, V1N 1K6, Canada
stbennet@wkpowerlink.com

P. Sherrington

RR #2, Cochrane, AB, T0L 0W0, Canada

L. Wilkinson

Wildlife/Habitat Inventory Specialist
British Columbia Ministry of Environment, Lands and Park
Room 400, 10003 110th Ave., Fort St John, BC, V1J 6M7, Canada

ABSTRACT

Little is known about the status of the northern goshawk (*Accipiter gentilis*) in northeastern British Columbia due to the remoteness and limited accessibility of the area. However, current development activities require immediate input of species requirements for planning purposes. The primary objective of the inventory was to find and describe goshawk nest sites/territories. The study area was located in the Boreal White Black Spruce biogeoclimatic zone west and north of Fort Nelson. We surveyed habitats in 3 different ecoprovinces and associated ecoregions: the Northern Boreal Mountains (Muskwa Ranges, Muskwa Foothills), Boreal Plains (Sikanni-Beatton Plateau), and Taiga Plains (Fort Nelson Lowlands). All forest habitats were stratified into 4 seral stages (early, mid, mature and old) and 3 cover types (deciduous dominant, coniferous dominant and mixed). Two hundred and sixty call playback stations (CS) were surveyed in the summer of 1997. Three goshawks were detected (0.77 goshawks/100 CS). Seventeen standwatch surveys were conducted in March 1998. Three goshawks were detected. We also surveyed almost 110,000 ha using a fixed-wing aircraft and a helicopter (1997 and 1998), and found 110 stick nests. Most stick nests (77.8%) were in mature/old deciduous stands and 80% were in live trembling aspen trees (mean dbh 43 cm). Stick nests were easily detectable from fixed-wing aircraft in March. The rate of goshawk detections and the structure of the stick nests we assessed were comparable to the results of other studies. However, the results of the call playback and nest surveys were somewhat contradictory, demonstrating that more work needs to be conducted on goshawks in northern British Columbia.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Demographic Performance and Microhabitat Variability in a Threatened Species, *Calochortus lyallii*[1]

Michael T. Miller
Department of Biology, University of Victoria
Victoria, BC, V8W 3N5, Canada

Key words: *Calochortus lyallii*, elasticity analysis, grassland, Liliaceae, Lyall's mariposa lily, pollination.

EXPANDED ABSTRACT

Little is known about the factors constraining the dynamics of peripheral populations, but such information may be critical to the management of rare species. My research focuses on long-term demographic variability and its relationship to spatio-temporal environmental factors in a restricted grassland endemic, Lyall's mariposa (*Calochortus lyallii*). Lyall's mariposa is a long-lived bulbiferous perennial in the lily family. It reaches its northern range limit in south-central British Columbia, where it is represented by only a few known colonies on a single height of land beside the U.S. border, south of Richter Pass. This area (known locally as Black Mountain) has been subject to various disturbances in recent years, including timber harvesting, livestock grazing, tree planting, and invasion by exotic weeds. Due to its vulnerability to disruption from both stochastic environmental events and human encroachment, *Calochortus lyallii* has been designated as threatened by the British Columbia Conservation Data Centre and is also a candidate for federal threatened status.

Endemism and rarity are unusually prevalent in *Calochortus*, where about half of the 60 species occurring in North America are listed as threatened or endangered. My objectives are (1) to identify those life history stages most critical to population growth and persistence in *Calochortus lyallii*; (2) to determine whether the relative importance of these stages varies in space and time; and (3) to compare the local population ecology of this species with that of a widespread, sympatric congener, *C. macrocarpus*. In so doing, I hope to gain a clearer insight into the causes and consequences of rarity, and to highlight specific life history and demographic characteristics that may distinguish rare organisms from more common ones.

Results pertaining to objective (3) are not dealt with here. To address objectives (1) and (2), annual censuses of some 1,300 *Calochortus lyallii* individuals at 3 separate localities representing a range of densities and microhabitat conditions were conducted from 1996 to 1998, and reproductive effort and population growth rates were compared between sites and years.

Lyall's mariposa typically occurs on well-drained, shallow soil in grassy forest openings, but sites have varying slope, aspect, moisture regime, litter depth, and associated plant cover. Stage-based projection matrix analyses for *Calochortus lyallii* show that life-stage distribution and projected population growth rates (λ) vary among sites and also between years, as do fruit set and total seed production. A small solitary bee (Halictidae: genus *Duforea*) was a frequent pollinator of *Calochortus lyallii* in 1996, was absent from all observed populations in 1997, and reappeared in 1998. Lower rates of fruit set in 1997 may reflect the absence of the pollinator during that year. These are the first known observations of *Calochortus lyallii*'s primary pollinator.

Elasticity analysis of the matrices suggests that the life history stages that have the greatest influence on λ are those concerned with survival and growth of vegetative plants, while fecundity has the lowest elasticity. However, elasticities were themselves sensitive to the structure of the original transition matrix, and changed in relative value depending on whether a particular population was projected to grow or shrink. At the same time, the seeds, which are gravity dispersed, tend to fall close to the parent plant, and it may be the inability to disperse to unoccupied sites, rather than the ability to perform above some demographic threshold, that limits the local distribution of this species in British Columbia. If so, preservation of unoccupied but potential *Calochortus lyallii* habitat on Black Mountain and further north will become increasingly important as original sites begin to be compromised by disturbance.

[1] Note: This paper summarizes the author's Ph.D. research. For a more detailed description of the ecology and status of *Calochortus*, see: Miller, B. T., and G. W. Douglas. Submitted. Status of Lyall's mariposa lily, *Calochortus lyallii* (Liliaceae), in Canada. Can. Field-Nat.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

A Range-Wide Evaluation of the Population Dynamics and Ecological Factors Associated With Bighorn Sheep Dieoffs

R.J. Monello
Department of Fish and Wildlife Resources
University of Idaho, Moscow, ID 83844 USA
mone9505@novell.uidaho.edu

D.L. Murray
Department of Fish and Wildlife Resources
University of Idaho, Moscow, ID 83844 USA

E. F. Cassirer
Idaho Department of Fish and Game
150 Warner Ave., Lewiston, ID 83501 USA

ABSTRACT

Bighorn Sheep (*Ovis canadensis*) populations throughout much of western North America have shown irregular cycles of abundance characterized by periods of rapid growth followed by drastic declines. The proximate cause in the majority of dieoffs is attributed to the pathogen *Pasteurella* (spp.), which causes pneumonia and death in infected animals. Pneumonia outbreaks may be a consequence of pathogen transmission from domestic sheep or goats to bighorns, but outbreaks may also occur in the apparent absence of bighorn-livestock interaction. Causal agents can be pathogens commonly present in animals with no clinical signs of pneumonia, suggesting that the disease is predisposed by certain ecological or environmental conditions. This project examines factors potentially predisposing bighorn sheep to population declines by examining the correlation between bighorn population trends and environmental and biological factors. We collected individual herd composition, herd history, precipitation, and temperature data for over 150 sheep populations ranging from the southwestern United States to Alaska. For each herd, we gathered all available demographic data and conducted interviews with state and provincial biologists to determine: 1) the cause, severity, and length of dieoff; and 2) the distance of each herd to neighbouring bighorn herds, domestic sheep, and other livestock. Results from 142 United States. herds indicate 61 (43%) have undergone a 17-90% decline in a single year. Of these, 32 (52%) dieoffs were likely due to *Pasteurella* (spp.), 12 (20%) to a variety of causes (e.g., mountain lion predation, scabies, and drought, etc.), and 17 (28%) declined because of unknown causes. The most common mode of disease transmission was from a neighbouring bighorn herd, usually infected by prior contact with domestic sheep or feral goats. Dieoffs always occurred during herd population peaks, suggesting an increased susceptibility to *Pasteurella* (spp.) caused by inadequate nutrition, increased dispersal of rams, or other density-dependent factors. Management implications and the ecological and environmental correlates to bighorn dieoffs will be discussed.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

The Effect of Grazing on Ant Biodiversity in the South Okanagan

Jennifer Heron

Department of Zoology, University of British Columbia
6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada
jmheron@interchange.ubc.ca

ABSTRACT

The antelope-brush ecosystem of British Columbia's South Okanagan region is one of the 4 most endangered ecosystems in Canada. Approximately 9% of this ecosystem remains relatively undisturbed. Widespread livestock grazing is 1 of the major disturbances to this ecosystem's community of arthropods, including ants. The effects of grazing on ant biodiversity are relatively unknown, partly owing to the lack of research on the ant fauna of British Columbia. The objective of this study is to compile an up-to-date list of ant species within this antelope-brush ecosystem, and to determine the effects of grazing on ant biodiversity. Approximately 95,000 ants were collected by pitfall trapping at 10 study sites over a 2-year study period (May 1994–October 1995). Each study site was classified as ungrazed, moderately grazed, or heavily grazed. Average catch per trap over this collection period was highest in the heavily grazed sites. Identification of ants from 1 collection period (June–July 1995) shows that the highest species diversity occurred in the ungrazed sites. These preliminary results suggest excessive grazing may reduce ant biodiversity in this ecosystem.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

Harlequin Duck Breeding Distribution and Hydroelectric Operations on the Bridge River, British Columbia

Edward L. Hill
BC Hydro
6911 Southpoint Drive, Burnaby, BC, V3N 4X8, Canada
ed.hill@bchydro.bc.ca

Kenneth G. Wright
Harlequin Conservation Society
Site 12 Comp 15, Galiano Island, BC, V0N 1P0, Canada

ABSTRACT

Studies about impacts of hydroelectric operations on riverine-dependent birds are limited to date. We present data on the distribution of a breeding population of harlequin ducks (*Histrionicus histrionicus*) on a river system in the Southern Interior of British Columbia that has been developed for hydroelectric generation. Pair surveys revealed 17 ± 5 (SD) birds occurring on the Bridge and Yalakom rivers, downstream of Terzaghi Dam. Four broods were observed in the lower Bridge River below Terzaghi Dam in 1997 and 1998. Mean brood size was 2.25 ducklings ± 1.26 (SD) in 1997 and 2.0 ± 1.15 in 1998. Only 2 broods were observed on the upper Bridge River. Physical habitat features are described. Our preliminary data indicate that harlequin ducks react to habitat changes resulting from hydroelectric operations. Increased flows below Terzaghi Dam in May, June, and July may reduce harlequin duck productivity.

Key words: breeding distribution, Bridge River, habitat characteristics, harlequin duck, *Histrionicus histrionicus*, hydroelectric operations, impact assessment, population.

Harlequin ducks (*Histrionicus histrionicus*) are known mostly from their coastal haunts where they inhabit rocky intertidal shorelines (Palmer 1976). Breeding occurs primarily along fast flowing rivers and streams (American Ornithologists' Union 1983). Presently, only scant information exists on the breeding abundance and distribution of harlequin ducks in British Columbia (Campbell et al. 1990, Wright and Goudie 1998). Virtually no information exists on the impacts of hydroelectric development or operations on harlequin ducks or mitigative measures. Alteration of rivers by hydroelectric dams and other anthropogenic disturbances threaten the globally endangered torrent duck (*Merganetta armata*; Callaghan 1997), which occupies the ecological niche of the harlequin duck in South America. In this paper, we examine nesting phenology, distribution and the physical habitat characteristics of a breeding population of harlequin ducks that occur within a hydroelectric system.

The Bridge River Hydroelectric System, located about 200 km northeast of Vancouver, includes Downton Reservoir and the La Joie Generating Station; Carpenter Reservoir and the 2 Bridge River Generating Stations on the northwestern shore of Seton Lake; and the Seton Generating Station on the west bank of the Fraser River near Lillooet. Hydroelectric development in the Bridge River System began in the 1920s.

Carpenter Reservoir was impounded in 1948. Terzaghi Dam, located about 41 km upstream from the confluence of the Bridge and Fraser rivers, diverts water into Seton Lake from Carpenter Reservoir. Consequently, the river channel is normally dry for approximately 4 km below the dam. Flows in the lower Bridge River are maintained by groundwater and tributary inflows. Water is rarely released directly into the Bridge River at Terzaghi Dam, except during the spills for testing or reservoir management. Spills have occurred 17 times from 1960 to 1997 (flow range during spills: 0.5–347.6 m^3/s).

Normally, water is not discharged past Terzaghi Dam. However, inflows to the Bridge River System were higher than average in May 1997 because of an increased snowpack and precipitation. This resulted in the need to manage water levels in the Downton, Carpenter, and Seton Reservoirs to minimize environmental impacts. With the agreement of regulatory agencies and other stakeholders, a controlled spill occurred at Terzaghi Dam from 13 July to 2 September 1997, with a peak discharge of approximately 27 m^3/s. Prior to initiating the spill, an environmental management plan (British Columbia Hydro 1997) was prepared to assess potential environmental impacts, and to identify possible mitigative and monitoring activities. The harlequin duck was identified as a species of concern, and inventory-level surveys in 1997 confirmed a breeding population using the Bridge River System.

In 1998 additional information on population size, habitat utilization, and nesting biology of harlequins in the Bridge River System was obtained to better define operational impacts and possible mitigative measures. Because no spill

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb ,1999. Volume One. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 490pp.

occurred at Terzaghi Dam in 1998, we were able to collect baseline data below the dam. Control data were also collected from the unregulated upper Bridge River which flows into Downton Reservoir.

Before the 1997 harlequin duck study (Wright 1998), little information was available on wildlife within the Bridge River watershed aside from anecdotal observations. We now know that the Bridge River supports a rich riverine bird community. The Bridge River also supports populations of several resident and anadromous fish species (Riley et al. 1998).

In the past, spills of water at Terzaghi Dam caused conflicts between the management of water and fisheries. Recognizing the need to collaborate on existing programs and/or initiate new fisheries improvement programs as well as to responsibly manage spills, British Columbia (BC) Hydro, the Stl'atl'imx Nation, the Department of Fisheries and Oceans, and the Ministry of Environment, Lands and Parks have agreed to implement the Bridge Seton Interim Cooperative Fisheries Project, and will shortly be implementing a Water Use Plan. The fisheries project is interesting for wildlife perspective because it includes the following components:

1. the design and construction of a water release structure at Terzaghi Dam;
2. the development and implementation of a water release strategy by 1 August 1999;
3. habitat improvement in the 4 km of dry river channel immediately below Terzaghi Dam; and
4. the Bridge River aquatic monitoring program (adaptive management).

The release of water from Terzaghi Dam (a fisheries-related commitment), and the subsequent permanent wetting of the presently dry channel will affect the ecology of riverine wildlife species such as harlequin ducks. Wright (1998) observed 1 brood of harlequins in the upstream section of the lower Bridge River that was wetted by the 1997 spill. Determination of the appropriate levels of flow below Terzaghi Dam will allow evaluation of the impacts of various flow regimes on harlequins and other wildlife, and collaboration with fisheries researchers on the interaction between fish and wildlife (e.g., predation) during the flow release experiment. This work will also be useful for wildlife-sensitive water use planning in other watersheds.

STUDY AREA

This study was conducted within the Bridge River watershed (a tributary of the Fraser River), British Columbia (Fig. 1). The Bridge River originates from a glacier in the Coast Mountain Range about 50 km west of Gold Bridge.

The upper Bridge River includes the Interior Douglas-fir (900–1100 m) and Montane Spruce (>1100 m) biogeoclimatic zones in its lower reaches. At higher elevations and westward, precipitation (mainly snow) increases, and the Engelmann Spruce–Subalpine Fir zone dominates. Most of the lower Bridge River occurs in the hot and dry (rainshadow conditions) Ponderosa Pine zone, whereas the Yalakom River and the Bridge River upstream of Hell Creek are within the Interior Douglas-fir zone.

Our surveys in 1998 were conducted on the upper Bridge River from Downton Reservoir to the glacier, the lower Bridge River from the Fraser River confluence to the upstream limit of water (about 4 km below Terzaghi Dam), and the Yalakom River from its confluence with the lower Bridge River to upstream of Shulaps Creek (Figs. 2, 3).

METHODS

Breeding Population, Phenology, and Distribution

We inventoried harlequin ducks by helicopter and by hiking surveys using techniques proposed by the Resources Inventory Committee (Resources Inventory Committee 1997).

Two methods were used to estimate the breeding population: (1) a pair of replicate surveys conducted 10 days apart during the prenesting period, and (2) capture–mark–recapture (CMR). The CMR method is based on Chapman (1951) with recent revisions (Seber 1977), and is generally the preferred method in a closed population. Birds were captured by flushing them into a mist net strung across the river, and were subsequently banded and released.

Approximate timing of nest initiation was investigated by comparing the frequency of single males to total birds observed. Chi-square goodness-of-fit tests with $P = 0.05$ significance level were used for this comparison. We aged broods according to Gollop and Marshall (1954), with modifications (Kuchel 1977). Back-dating provided approximate dates of clutch initiation and hatching. Fledging dates were estimated from brood age.

Locations of harlequin sightings during pair (prenesting; May–June) and brood (July–August) periods were plotted on field maps and are discussed in the following section.

Habitat Characteristics

All sites where ground observations were made were described. We recorded the following variables: bank type (vertical or horizontal), bottom type, channel form, bank composition, vegetative cover, water clarity, loafing site availability, channel width, stream width, gradient, and depth. Loafing sites are usually rocks and instream islands surrounded by moving water (Bengtson 1972); however, harlequins will also use logs and stream banks in some situations. All sites where birds were observed during ground surveys were photographically documented; stream gradient was measured using a clinometer; channel widths were estimated visually; and maximum water depth was recorded. We also described some general physical habitat attributes of

sites occupied during the pair and brood cycles and at randomly chosen sites throughout the study area.

RESULTS

Breeding Population, Phenology, and Distribution

Lower Bridge and Yalakom Rivers

An estimated breeding population of 17 ± 5.5 (SD) birds in 1998 was found on the lower Bridge and Yalakom rivers. This was similar to our maximum count for the same area (15 individuals, including 5 pairs).

The proportion of single males to total birds observed on the Yalakom and lower Bridge rivers did not differ significantly between hiking surveys on 3–6 May and 14–17 May ($\chi^2_1 = 0.741$, $P > 0.25$, Yates corrected). This result indicates that laying was not completed by mid-May 1998.

During the prenesting period, most pairs were observed on the Yalakom River and at its confluence with the lower Bridge River (Fig. 3). Apart from birds in the Bridge River/Yalakom River confluence area, very little use of the lower Bridge River was observed. No birds were observed downstream of the confluence of the lower Bridge and Yalakom rivers in either year, except a broodless female on 17 July 1998. Nesting on the lower Bridge and Yalakom rivers occurred from early May to late June (Table 1).

Four broods were documented on the lower Bridge River in both years in the reach upstream of the Yalakom River. Mean brood sizes were 2.25 ± 1.26 (SD) in 1997 (spill year) and 2.0 ± 1.15 in 1998 (normal year), and did not differ significantly ($U = 7$, $P > 0.05$; Mann-Whitney U-test). Broods in 1997 were distributed closer to Terzaghi Dam than in 1998, presumably because of the increased flow. We observed little movement of known broods (maximum 4 stream km). No broods were found using the Yalakom River in 1997 or 1998.

We banded 15 harlequin ducks throughout the 1998 season (adults = 7, young of year = 8). A male banded on the lower Bridge River on 3 May was subsequently resighted alone on 16 May on the Yalakom River, 6 km above the lower Bridge River confluence. Of the 3 females banded during the prenesting period, 2 were later recaptured with young on the lower Bridge

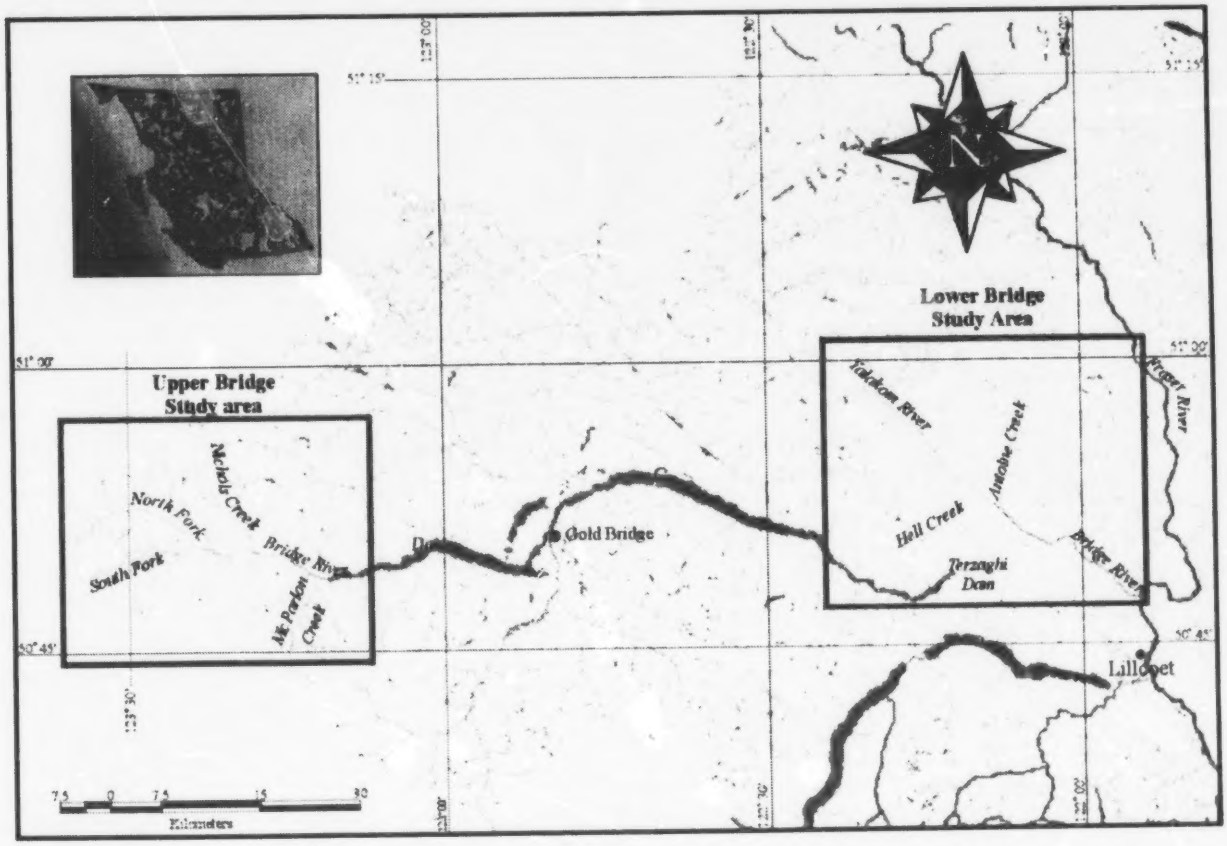

Figure 1. Study area map of the Bridge River System, British Columbia.

River, only a few kilometres upstream of the pair banding site.

Three birds previously banded elsewhere were resighted in the Yalakom River and lower Bridge River study area, linking this breeding population with coastal wintering areas in British Columbia and Washington State. These birds consisted of a female and male banded at coastal moulting areas in the Strait of Georgia in 1994–1996, and a male that moulted at Protection Island, Washington, in 1994 (G. Schirato, Washington Dep. Fish and Wildl., 1988, pers. comm.).

Upper Bridge River

A CMR estimate was not possible on the upper Bridge River due to high water levels. A maximum count of 24 birds (11 pairs and 2 single males) was obtained for the upper Bridge River on the 13 May 1998 helicopter survey. Pairs were distributed relatively evenly from McParlon Creek upstream to the confluence of the forks (Fig. 2). The count from the 2 June 1998 helicopter survey was significantly lower (i.e., 7 birds; $\chi^2_1 = 9.36$, $P < 0.005$). This observed decrease was most likely attributable to the onset of nesting. We noted a significant difference in the proportion of single males observed between the 13 May helicopter survey and the 3–7 June hiking survey ($\chi^2_1 = 14.98$, $P < 0.01$), indicating that nesting began in late May–early June 1998.

Only 2 broods were observed on the upper Bridge River (mean brood size 2.5 ± 0.71) which precluded discussion on distribution. High water levels and increased turbidity may have caused broods to move into clear water tributaries of the upper Bridge River. The upper Bridge River is now considered an unsuitable control site for the lower Bridge River due to habitat differences (higher elevation, reduced sightability, significantly higher flows).

Habitat Characteristics

Twenty-four occupied sites and 10 random sites were assessed. Channel gradient was consistently higher at occupied

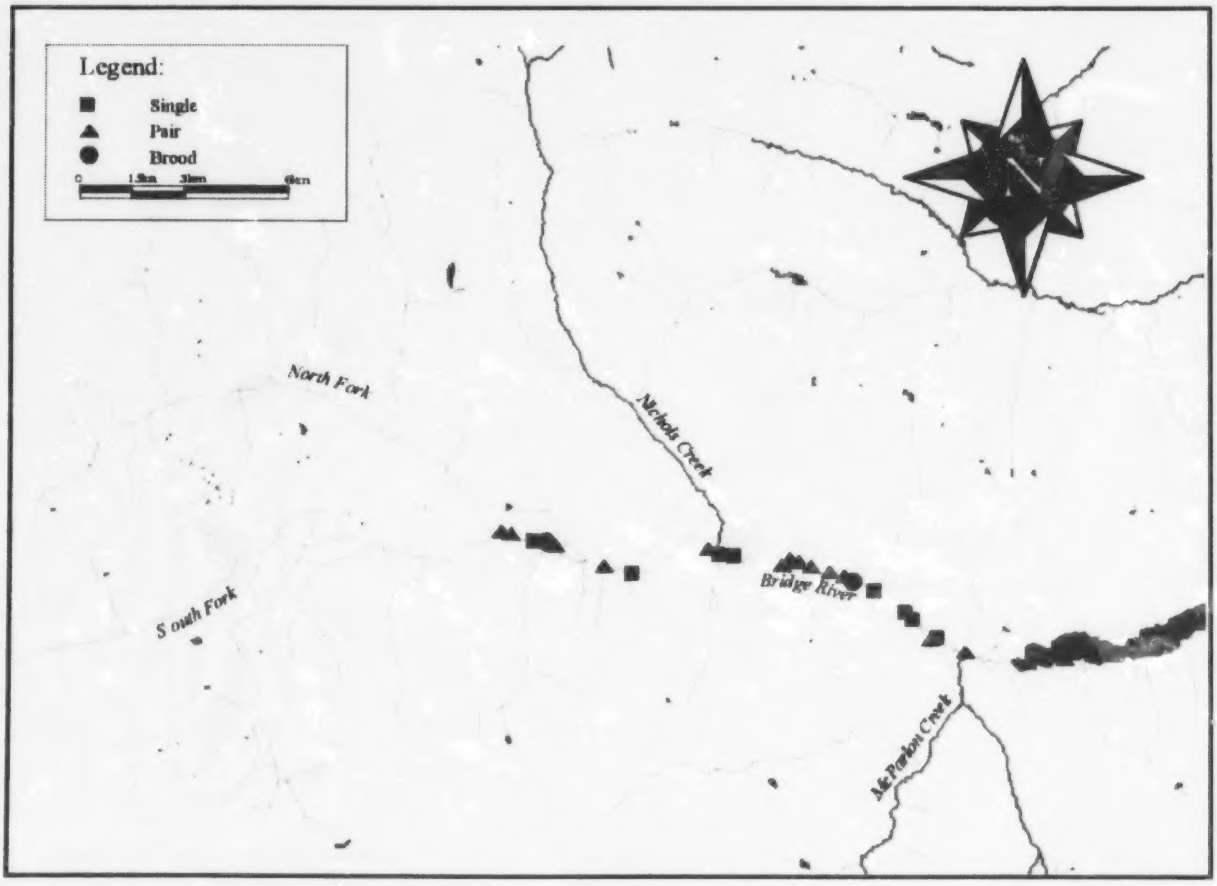

Figure 2. Distribution of harlequin ducks (May–July 1998) on the upper Bridge River, British Columbia.

sites than random sites (mean = 2.3% ± 1.7 SD and 1.6% ± 0.9, respectively). Water clarity was variable and reach specific. The upper Bridge River was turbid with glacial till during the pair/prenesting period while the lower Bridge River above the Yalakom River was clear. The Yalakom River, and the Bridge River downstream of the Yalakom River confluence were moderately turbid. Channel width at sites occupied by harlequin ducks was wider on the lower Bridge River (14.9 m ± 5.2) than the Yalakom River (10.0 m ± 3.0); however, random sites barely differed. Stream width was similar among occupied and random sites. Bottom substrate mostly consisted of cobbles with some sites having predominately boulders. Harlequins

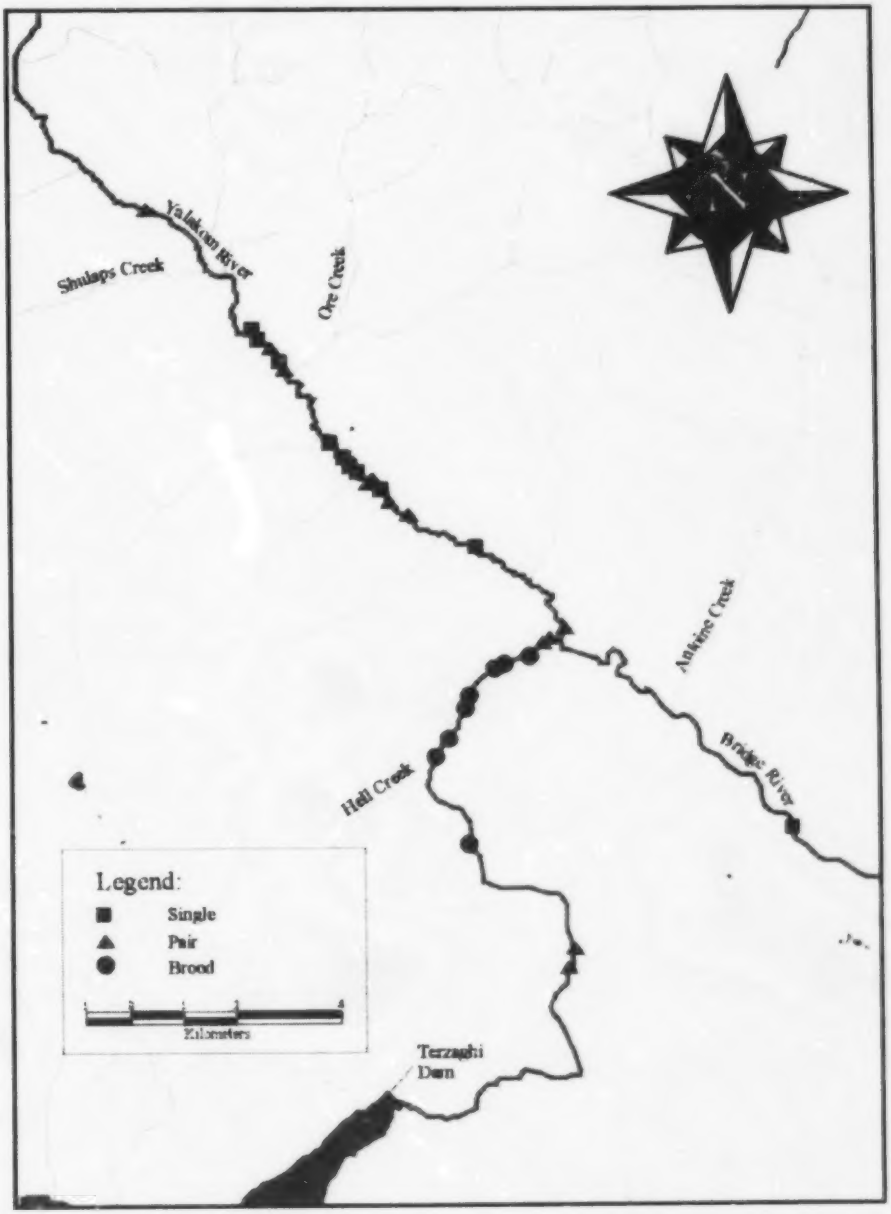

Figure 3. Distribution of harlequin ducks (May–August 1998) on the lower Bridge and Yalakom rivers, British Columbia.

Table 1. Brood class, age, and breeding phenology of harlequin ducks captured July–August 1998, lower Bridge River, British Columbia.

Date banded	Brood size	Class[a]	Age (days)	Estimated hatch date	Clutch initiation[b]	Fledgling[c]
18 July	3	IIb	26–38	10–22 June	1–13 May	25 Jul–6 Aug
20 July	1	IIb	26–38	12–26 June	3–15 May	27 Jul–8 Aug
2 Aug	1	IIa	21–25	8–12 July	29 May–2 June	22–26 Aug
3 Aug	3	IIa	21–25	10–14 July	3 May–3 June	23–27 Aug

[a] Follows Gollop and Marshall (1954) as modified by Kuchel (1977).

[b] Estimated by backdating from estimated hatch date, allowing 28 days for incubation and a 6 egg clutch with a laying interval of 1 egg /2 days (Bengtson 1972).

[c] Estimated as 45 days from hatching.

showed a preference for areas with numerous instream boulders and logs for loafing.

DISCUSSION

Little information presently exists on harlequin duck breeding populations in British Columbia (Campbell et al. 1990, Wright and Goudie 1998). Therefore, it is difficult to identify significant breeding concentrations and any regional patterns of distribution. In much of its range, the harlequin duck exhibits low breeding densities. Populations have been documented in the Canadian Rockies: 41 ± 8, Kananaskis River, Alberta, Smith 1999; 150 ± 18, Bow River, Alberta, Smith 1998; 58 ± 7, McLeod River/Whitehorse Creek System, Alberta, MacCallum 1997. The upper Bridge River seems to support a significant breeding population with 11 pairs and 2 single males (24 individuals) observed on 13 May 1998, and may have supported twice that (i.e., 22 pairs or 48 birds) based on 50% sightability, as found on helicopter surveys conducted in Alaska in similar or better visibility conditions (McCaffery and Harwood 1996). The lower Bridge and Yalakom rivers, also support a relatively large breeding population (17 ± 5 birds). In the larger Nahatlatch River watershed west of the Fraser River near Boston Bar, a breeding population of 39 ± 8.9 birds was reported (Freeman and Goudie 1998). Estimates of density would be required to compare the reported populations because of differences in the sizes of study areas.

Harlequin ducks breed in both treeless (Bengston 1972) and forested areas. In the forested breeding range, harlequins show a preference for areas with an intact riparian reserve over logged stream banks (Freeman and Goudie 1998). They exhibit a preference for fast flowing (gradient 1–5%), shallow rivers and streams (Palmer 1976) that support available macro-invertebrates populations (e.g., larval stonefly *Plecoptera* spp.). Breeding streams in Alaska had significantly greater discharge volume, stream width, and riparian width than unoccupied streams (Crowley 1993). Rodway (1998) found that occupied streams were narrower than unoccupied streams, and had larger substrate and higher vegetation cover.

The results of the field work in 1997 and 1998, along with an inventory survey of Downton Reservoir in 1997 (Robertson et al. 1998), have substantially contributed to our knowledge of wildlife in the Bridge River System, and therefore to our understanding of the effects of hydroelectric operations on the system. A breeding population of harlequin ducks that uses the Bridge River below Terzaghi Dam has been documented. Furthermore, in 1997, a brood was observed using the section of river below the dam that was wetted by the controlled spill. In addition, 94 species of birds, 15 mammalian species, and 2 species of reptiles were documented.

Four broods were present in both years on the lower Bridge River upstream of the Yalakom River and average brood size was not significantly different between years. However, broods tended to occur closer to Terzaghi Dam in 1997 (during the spill) than in 1998. This difference in distribution probably resulted from increased water flows and possibly more favourable upstream habitat conditions.

MANAGEMENT IMPLICATIONS

The original objective of the harlequin studies in 1997 and 1998 was to obtain sufficient information to assess potential operational (controlled spill) impacts (e.g., direct mortality through loss of nests, drowning or habitat alteration, changes in productivity resulting from habitat alteration), and to identify possible mitigation and/or compensation (e.g., timing of controlled spills to avoid critical periods, habitat improvement). A staged approach to accomplish this objective was planned:

- determine presence of harlequins (and other species) in 1997;
- determine general habitat use and phenology in 1998;
- evaluate habitat use and phenology as directly related to BC Hydro's operations and modified flows (e.g., within spill channel) - future studies; and
- evaluate impacts/mitigation/compensation in cooperation with other stakeholders - future studies.

The original objectives have been partially met. We now

know that harlequins nest within the lower Bridge River System (probably along the Yalakom and other tributaries) and raise their broods on the main stem of the Bridge River. However, we do not know whether harlequins nest along the section of river immediately below Terzaghi Dam during spill years. Radiotelemetric studies are required to more accurately determine the nesting period, nesting locations, and habitat use by broods.

At present, we can say only that losses of nests and young broods may be reduced by keeping flows below Terzaghi Dam relatively constant from May through July.

The scope of proposed wildlife studies (particularly on harlequin ducks) has been altered by agreement on the Bridge Seton Interim Cooperative Fisheries Project and Water Use Planning. The decision to maintain a flow below Terzaghi Dam requires a broader scope to include wildlife in the instream flow needs assessment, planning of channel shaping, and adaptive management. Future work on harlequin ducks and other wildlife on the Bridge River will reflect these broader needs.

ACKNOWLEDGEMENTS

We thank the following individuals for their enthusiasm, expertise, and dedication on all aspects of the field work: S. Jackson, L. Campbell, J. Joy, K. Truman, R. Vennesland, J. Komaromi, and R. van den Driessche. P. Fraser (Cariboo-Chilcotin Helicopters) skillfully piloted on the aerial surveys. S. Hall kindly provided hospitality and assistance. M. Pepin prepared the illustrations. N. Baldwin and D. Loiselle assisted with the layout and poster presentation. I. Goudie reviewed an earlier draft. BC Hydro funded this study.

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Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999

Winter Responses of Forest Birds to Habitat Corridors and Gaps

Colleen Cassady St. Clair
Department of Biological Sciences, University of Alberta
Edmonton, AB, T6G 2H1, Canada
cstclair@ualberta.ca

Marc Bélisle
Centre de Recherche en Biologie Forestière
Université Laval
Québec, PQ, G1K 7P4, Canada

André Desrochers
Centre de Recherche en Biologie Forestière
Université Laval
Québec, PQ, G1K 7P4, Canada

Susan Hannon
Department of Biological Sciences
University of Alberta
Edmonton, AB, T6G 2H1, Canada

Key words: black-capped chickadee, corridor, downy woodpecker, forest fragmentation, gap, hairy woodpecker, *Picoides pubescens*, *Picoides villosus*, *Poecile atricapillus*, *Sitta carolinensis*, white-breasted nuthatch.

EXPANDED ABSTRACT

Forest fragmentation and habitat loss may disrupt the movement or dispersal of forest-dwelling birds. Despite much interest in the severity of these effects and ways of mitigating them, little is known about actual movement patterns in different habitat types. We studied the movement of wintering resident birds lured by playbacks of mobbing calls to compare the willingness of forest birds to travel various distances in continuous forest, along narrow corridors (fencerows), and across gaps in forest cover. We also quantified the willingness of black-capped chickadees (*Poecile atricapillus*) to cross gaps when alternative forested detour routes were available. All species were less likely to respond to the calls as distance increased to 200 m, though white-breasted nuthatches (*Sitta carolinensis*) and hairy woodpeckers (*Picoides villosus*) were generally less likely to respond than chickadees and downy woodpeckers (*P. pubescens*). Chickadees were as likely to travel in corridors as in continuous forest, but were less likely to cross gaps as gap distance increased. The other species were less willing to travel in corridors and gaps relative to forest, and the differences among habitats also increased with distance. For chickadees, gap-crossing decisions in the presence of forested detours varied over the range of distances we tested and were primarily influenced by detour efficiency (the length of the shortcut relative to the available detour). Over short distances, birds used forested detours regardless of their efficiency. As absolute distances increased, birds tended to employ larger shortcuts in the open when detour efficiency was low or initial distance in the open was high, but they limited their distance from the nearest forest edge to 25 m. Thus, chickadees were unwilling to cross gaps of >50 m when they had forested alternatives, yet they sometimes crossed gaps as large as 200 m when no such choice existed. Our results suggest that the presence of corridors enhanced the movement of some, but not all, forest birds, and that even chickadees, which were less sensitive to gap width, preferred not to venture far from forest cover. All text and figures from this poster contribution consisted of excerpts from St. Clair et al. (1998).

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A Summary of Roosting Requirements of Northern Long-Eared Myotis in Northeastern British Columbia

Maarten J. Vonhof
Echo Biological Consulting Inc.
1610 - 2A Street NW, Calgary, AB, T2M 2X4, Canada
mvonhof@yorku.ca

Lisa C. Wilkinson
British Columbia Ministry of Environment, Lands and Parks
Suite 400, 10003 - 110 Avenue, Fort St. John, BC, V1J 6M7, Canada

Key words: bat, boreal mixedwood forest, *Myotis septentrionalis*, northern long-eared myotis, roosting, wildlife tree.

EXECUTIVE SUMMARY

(Refer to Vonhof and Wilkinson 1999 for further details). Bats spend over half their lives subject to the selective pressures of their roost environment (Kunz 1982). Roosts offer protection from the ambient environment and from predators, provide sites for mating and rearing young, and promote social interactions. Despite the need for accurate information to include in habitat management plans, the roosting habitat requirements of forest-dwelling bats have only recently begun to receive attention.

Up to 7 species of bats are found in the boreal mixedwood forest in northeastern British Columbia, and all of these are dependent on forest habitat for their roosting and foraging needs (Nagorsen and Brigham 1993). In this summary we report on a 2-year study to examine the roost-site preferences of the northern long-eared myotis (*Myotis septentrionalis*), a provincially Blue-listed species. The specific objectives were: to characterize roosting habitat used by bats, with emphasis on northern long-eared bat females; to compare attributes of confirmed roost sites with potential roosts at the patch and stand levels to determine the criteria used for roost selection; and to provide specific recommendations as to how bat roosting habitat requirements can be incorporated into forest development plans and wildlife tree patch design.

The study area was located in northeastern British Columbia near Fort Nelson, within the moist-wet subzone (mw2) of the Boreal White and Black Spruce (BWBS) biogeoclimatic zone, in the Fort Nelson Lowland ecosection. The area was surveyed in the summers of 1997 and 1998. Bats were captured in mistnets, and radiotelemetry was used to locate roost sites. Various measurements were taken of the roost tree and a randomly selected cavity tree. A 17.8-m-radius plot was established around the roost tree and several site characteristics were recorded. A similar plot was established around a randomly selected cavity tree in another area of the same stand (as in Vonhof and Barclay 1996). Analyses were conducted to compare: various tree measurements; characteristics of cavity trees and non-cavity trees; tree characteristics of roost trees and cavity trees from the 2 geographic scales (cavity trees within the roost tree plot versus cavity trees from other areas of the same stand); and site characteristics of roost trees and cavity trees from the 2 geographic scales.

Nine female and 4 male northern long-eared myotis were outfitted with radio-transmitters, yielding 21 and 17 roost trees respectively. Females roosted primarily in cracks (17), followed by primary cavity excavator hollows (3), and beneath loose bark (1), and males roosted primarily beneath loose bark. Females typically roosted in colonies, whereas all males roosted alone. Female roost trees were farther away from the nearest neighbouring tree than were cavity trees from the same forest patch. No tree or site characteristics significantly discriminated between roost trees and cavity trees in other areas of the same stand. Females roosted at random with respect to the availability of tree species and decay stages, although all roosts were decay stage 2 trembling aspen or balsam poplar trees, and were located in age class 6 (mature) stands. Almost all cavity trees measured were deciduous. Bats commonly switched roosts each day, even when they were caring for nonvolant young, and the distance between roost trees was small (approximately 200 m on average).

Our results concur with other studies, which found that northern long-eared myotis roost primarily in mature stands of deciduous trees, which contain a greater abundance of large snags in a variety of decay classes (Cline et al. 1980), reduced densities, and less clutter (Franklin et al. 1981). In terms of management recommendations, the presence of cavity trees should be considered when designating wildlife trees; all lactating females used cracks, which provide both security and thermal cover. In addition, harvesting plans should consider leaving veteran and dominant trees to help maintain some degree of canopy cover and associated microclimate; by leaving both live and dead trees, a range of decay classes as well as cavities and loose bark will be preserved. Selective cutting could also help to reduce the level

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of clutter in dense second-growth stands. Lastly, because bats use a suite of roosts within a restricted area, roosting habitat could be provided by maintaining contiguous stands or large patches of mature deciduous or mixedwood forest.

The recent interest in mixedwood harvesting in northeastern British Columbia makes the results of this study and the accompanying management recommendations highly relevant to the conservation of habitat for northern long-eared bats, as well as for other bat species and cavity users.

ACKNOWLEDGEMENTS

This study was funded by Forest Renewal British Columbia. Several people provided valuable assistance in the field: S. Bradbury, S. McNally, J. Unger, and A. Yee. S. Morris provided information on netting sites, and G. Haines and T. Li provided GIS support. The British Columbia Ministry of Environment, Lands and Parks in Fort St. John provided telemetry equipment and a GPS unit.

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At Risk

VOLUME TWO

Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk

University College of the Cariboo
Kamloops, BC, February 15-19, 1988

Edited by Laura M. Darling

British Columbia Ministry of Environment, Lands & Parks
University College of the Cariboo

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February 15 - 19, 1999
University College of the Cariboo
Kamloops, British Columbia

edited by
Laura M. Darling
Wildlife Branch
British Columbia Ministry of Environment, Lands and Parks
Victoria, British Columbia

August 2000

SPONSORS OF THE PUBLICATION

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Canadian Cataloguing in Publication Data
Main entry under title:
At risk: proceedings of a conference on the biology and management of species and habitats at risk.
Includes bibliographical references.
ISBN 0-7726-4348-2

1. Endangered species - British Columbia - Congresses.
2. Biological diversity conservation - British Columbia - Congresses.
3. Wildlife habitat improvement - British Columbia - Congresses.
I. Darling, Laura M. II. University College of the Cariboo. III. British Columbia. Ministry of Environment, Lands & Parks.
QH77.C3A8 2000 333.95'22'09711 C00-960290-9

Cover artwork by Michael Hames

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PREFACE

This collection of papers stems from the third conference in British Columbia to focus on species and habitats at risk. The first, co-hosted by the Federation of British Columbia Naturalists, Douglas College, and the British Columbia Ministry of Environment was held at Douglas College in Vancouver in 1980 (Stace-Smith et al. [eds.], 1980, *Threatened and Endangered Species and Habitats in British Columbia and the Yukon*). The second was sponsored by the Federation of British Columbia Naturalists and the Northwest Wildlife Preservation Society and was held in Vancouver in 1991 (Rautio [ed.], 1991, *Community Action for Endangered Species*). This most recent meeting, held in Kamloops, 15 – 19 February 1999, and co-hosted by the British Columbia Ministry of Environment, Lands and Parks and the University College of the Cariboo, had more than 60 partnering organizations and businesses. Interest and participation in each of these conferences was high and attendance was at capacity. The papers in the proceedings from these 3 conferences can be used to gauge of the state of our knowledge of the biology of species. At each of the first 2 conferences, 8 papers dealt with the biology of a species at risk. In this third set of proceedings, there are papers dealing with aspects of the biology of more than 60 species. Some of the titles of the papers presented in Kamloops would have been unimaginable at the earlier conferences—for example, an overview of research on the marbled murrelet would have been a very slim paper a decade or 2 ago!

The proceedings from the Kamloops conference also hint at other emerging trends in approaches to species and habitats at risk. In the past decade, increasing concern about loss of biodiversity and the growing number of species at risk has resulted in an increase in research and inventory work, especially for species groups and habitats that have been traditionally ignored by wildlife management. As a result, included in these proceedings are preliminary lists of species at risk for mosses, marine invertebrates, dragonflies, beetles, freshwater molluscs, and butterflies. "Natural history" has also resurfaced as an important field of study as such information becomes crucial to the proper management of species, particularly those at risk, and this renaissance is reflected in a number of papers. An expanding awareness of the role of organisms at risk in ecological processes is highlighted in several places in these proceedings. Examples include discussions of the fragile cryptogamic crusts of grasslands and shrub-steppes, and of the complicated and fragile interconnections of mycorrhizal fungi.

The Kamloops conference and its proceedings are an important step in communicating the results of the recent, original and innovative work to managers, scientists, educators, naturalists, and the general public. There are many sobering signs of how desperate the situation has become: far more extensive and complicated endangered species lists; a growing number of ecosystems at risk; an increasing understanding of how fragile some ecosystems are; and accelerated rates of habitat loss and fragmentation. On the bright side, there is the tremendous energy that has in recent years been put toward public education, and documenting, classifying, researching, and conserving species at risk.

These papers are a valuable glimpse into that effort.

David Fraser and Karl Larsen, Co-chairs

Eric Lofroth, Don Reid, Laura Darling, and Eric Parkinson, Organizing Committee

EDITOR'S NOTE

The papers in these Proceedings have not been peer reviewed – the content is fully the authors' responsibility. Each paper was subjected to a grammar and copy edit, to address issues of spelling, grammar, and conventions of style and layout selected for this document. I made it my responsibility to review all edit suggestions made by our copy editors, and to approve all changes I felt did not alter the intent of the author; most authors were contacted during this process to address contentious edits, to clarify wording, or to provide missing documentation. The papers were submitted at varying degrees of technical detail, and individual writing styles varied substantially; however, we made no attempt to change the character of the papers submitted. Graphics were submitted in numerous production formats and layout styles; we have done our best to ensure that each graphic is readable and gets the point of the graphic to the reader. The final product is the result of the immense effort by many people to make each paper better.

LAURA M. DARLING, EDITOR

ACKNOWLEDGEMENTS

Many hands make light work. These Proceedings and the conference from which they stem could not have happened without the input and support of many individuals and organizations.

The conference, sponsored by the B.C. Ministry of Environment, Lands and Parks and the University College of the Cariboo, was supported by financial and in-kind contributions from over 25 agencies and organizations, whose logos are presented on page iii of this volume.

Publication of these Proceedings was supported by financial contributions from 9 agencies and organizations whose logos appear on page ii. Special thanks to BC Hydro for making a substantial cash contribution, and to Ed Hill for making it happen. Authors of lengthy papers were called upon to contribute 'page charges' to cover costs of editing and layout. Additional financial support was provided by John Carnio, Curator of Mammals, The Toronto Zoo. Many thanks to Queen's Printer for rock-bottom pricing for print production, facilitating greater distribution of these documents.

The proceeds of a silent auction held at the conference also were directed towards the publication of these proceedings. The following organizations and individuals generously donated items: The Nature Trust of British Columbia, Nature Conservancy Canada, Ian Goudie, Bear Brewing Company, Lone Pine Press, Northwest Wildlife Preservation Society, Fraser's Thimble Farms, David Fraser, Neville Crosby Inc., Eric Lofroth, Vancouver Aquarium, Fred Cooke, Western Canada Wilderness Committee, Goldstream Park Nature House, Columbia Basin Fish and Wildlife Compensation Program, Greystone Press, Mountain Equipment Coop, UBC Press, Parks Canada, Wild Birds Unlimited, B.C. Conservation Data Centre, Vancouver Telescope Center, Royal B.C. Museum, Habitat Acquisition Trust, Creston Valley Wildlife Management Area, Province of B.C. Wildlife Tree Committee, David Green, The Field Naturalist, Dick Cannings, Rob Butler, Vancouver Island Marmot Foundation, BC Hydro, The Steelhead Society, Laskeek Bay Conservation Society, Robert Busch, Canadian Forest Products, World Wildlife Fund, Liz Hammond-Kaarremaa, Friends of Ecological Reserves, Jim Pojar, International Pacific Halibut Commission, Crown Publications, Tracee Geernaert, Larry Halvorsen, Bill Harper, and John Krebs.

Thanks to those who ensured the conference ran smoothly: J. A. Wightman for fund raising, conference planning, and administrative management of incoming submissions and conference scheduling; T. Rankin of the Forestry Continuing Studies Network for initial organization of the conference; G. Lyons and the UCC Activity Centre staff who orchestrated minute-to-minute activities at the UCC Campus Activity Centre; A. Harcombe and D. Eastman for input to selection of papers; B. Gowans, for her part in the silent auction; Kamloops Naturalist Club for running field trips at the conference; D. Lewis and R. Morris, for help in preparing conference packages; D. Williams, for organizing the poster session at the conference; session chairpersons for keeping speakers on time; T. Dickinson , B. Heise and W. Klenner, for hosting the keynote speaker and public lecturers; 'Bookies' and 'Strictly For The Birds' for acting as ticket agents for the public lectures; Roger Barnsley (President of UCC), John Jules (Kamloops Indian Band), and Cliff Branchflower (Mayor of Kamloops) for supporting the conference and graciously welcoming delegates; and Bear Brewery, for crafting a special run of 'Northern Spotted Ale' just for the conference.

C. Hoodicoff efficiently coordinated the huge volunteer workforce: N. Bertram, D. Bissonette, J. Bleuschke, A. Callahan, R. Canart, S. Cote, M. Grant, D. Haughland, T. Hoodicoff, J. Hromadnik, K. Johnston, C. Kissick, L. Larson, V. Larson, D. Lewis, K. Lowson, M. MacDonald, R. McAllister, R. Morris, C Nesbit, A. Norquay, J. Pedersen, K. Pedersen, D. Pettman, N Prichard, S. Raven, T. Richens, A. Ross, S. Rowsell, P. Ruley, K. Safford, L. Sheperd, J. Sorensen, B. Tanner, C. Tarasoff, A. Tipler, W. Tokarek, K. Turriff, S. Warman, and C. Yakowski.

Thanks to those who contributed to the production of these Proceedings: Ros Penty and Frances Backhouse, editors, and Larry Grainger, Wayne Sampson and Denton Pendergast of Rocket Science Design for patience, persistence and professionalism to improve the quality of each and every paper in this Proceedings; Gail Harcombe for facilitating and trouble-shooting production; UCC for carrying and administering proceeds from the conference; proof readers and other volunteers who ensured the final product was complete, M. Austin, M. Badry, C. Cadrin, S. Cannings, M. Chutter, S. Eaves, S. Flynn, L. Friis, L. Haggstrom, I. Hatter, J. Hobbs, G. Jones, D. Kang, N. Laity, T. Lea, E. Lofroth, Z. O'Neill, K. Paige, J. Quayle, L. Ramsay, L. Stanlake, H. Schwantje, L. Shaw, C. Tolkamp, B. von Sacken, L. Westereng, K. Wipond.

Thanks to Michael Hames for his artistic contribution to the conference and to species at risk; his art now graces posters, T-shirts and these Proceedings.

And a special recognition to each and every one of the presenters and delegates for participating.

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Life Beyond Salmon Streams: Communities of Headwaters and Their Role in Drainage Networks

John S. Richardson

Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks, and
Department of Forest Sciences, 2424 Main Mall, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada
jrichard@interchg.ubc.ca

ABSTRACT

The headwaters of our watersheds are important for a number of intrinsic reasons, as well as for their impact on maintenance of downstream environments. The emphasis of research and management in stream ecosystems has typically been on salmonid fish, to the neglect of other stream and riparian organisms. Headwaters are sources of a large proportion of the energy used to fuel river food webs via organic matter that enters headwaters in the form of leaf litter from riparian vegetation. Headwaters themselves harbour a number of poorly known species, some of which occur nowhere else. There are many species associated with these environments, especially invertebrates, for which we lack even the most basic of information. Finally, the cumulative effects of small, incremental alterations to headwater channels may have impacts on downstream environments, but we have yet to design studies that adequately address this issue.

Key words: cumulative effect, headwaters, productivity, riparian, stream.

Headwater streams serve important roles in drainage networks, including transport of water, sediment, and organic matter to lower reaches, in addition to providing a unique aquatic and riparian habitat. Since small streams make up much more than half of the total channel length in a watershed, the effects of land use can disturb the rates and quality of materials supplied to reaches downstream. These lower reaches are often habitat to fish and are thus given precedence for protection, rather than headwaters where other, lesser-known organisms are found.

Recent studies have demonstrated the critical role of organic matter input in supporting the productive capacity of streams (Richardson 1991, Wallace et al. 1997). Many species depend critically upon the energy provided by organic matter, by the habitat it creates, or both (Richardson 1991, 1992). Species diversity is, in part, a function of the productivity of a system. Forest harvest and other forms of riparian clearing obviously lead to severe reductions in organic matter inputs and shifts in community structure, but changes in communities can also be caused by changes in temperature patterns, algal growth, or hydrology. Exclusion of litter inputs or removal of leaf packs results in dramatic declines in diversity and productivity throughout stream food webs (Wallace et al. 1997, L. Rowe and J. Richardson unpubl. data). All of these studies point to the link between organic matter and the productivity base of the stream's food web, and to the importance of maintaining inputs of organic matter from riparian forests.

Small headwater streams are common, but their function in drainage networks and as habitat for organisms not associated with larger streams is poorly known. One of the main roles for riparian protection along larger streams is the supply of large woody debris, which may contribute habitat for some juvenile salmonids (Murphy and Koski 1989). We have little idea of the requirements for protection of headwater streams and whether changes in the tributaries may affect the downstream reaches that are provided with riparian reserves. Advances in the past decade make it possible to begin the task of evaluating the role of small streams in our watersheds.

In this paper I will discuss 3 themes related to the importance of headwaters for watersheds and for fish-bearing reaches downstream. First, organic matter supplied from the riparian zone is a critical and limited resource that forms the basis of most riverine food webs. This resource is supplied to downstream reaches from its input points, which are predominantly within headwaters. Second, headwaters and their riparian areas (including intermittent streams) provide habitat for many organisms of conservation concern, many of which we know very little about. Third, the issue of cumulative effects of changes to watersheds has been, and continues to be, difficult to address; however, this is a topic that needs to be studied.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

STUDY AREA AND METHODS

The small streams studied by my students and I have included streams in the University of British Columbia (UBC) Malcolm Knapp Research Forest, the Chilliwack River valley, throughout the coastal western hemlock biogeoclimatic zone, and in the Horsefly Forest District.

RESULTS

Organic Matter

Small streams may get more than 95% of the energy that fuels their food webs from organic matter (leaves, needles, branches) that falls into the stream channel from the riparian area. The input of this organic matter, or detritus, can limit the productive capacity of some streams (Cummins et al. 1989, Richardson 1993). Estimates of organic matter inputs, decomposition rates, and standing crops (Richardson 1992) lead to the conclusion that small streams lose 70–94% of their inputs of particulate organic matter (POM; >0.45µm) to downstream reaches. In more recent studies (P. Kiffney and J. Richardson unpubl. data) we have estimated POM outputs of small, second-growth streams to be 0.76 ± 0.17 kg/m of stream channel (ash-free dry weight ± 1 s.e.).

We used a series of simple calculations based on input rates of organic matter (approx 270 g/m²/yr) and stream channel areas (bankfull width X stream channel length ÷ 2 for the taper of the channel area). These particulate inputs were multiplied by a conservative estimate of lateral inputs (blow-in or wash-in from above the floodplain area) of 25%. Total inputs were discounted by 15% for losses due to leaching of soluble organics. Comparing our estimates of inputs and outputs would indicate that outputs are 4.79 (± 0.53) times that of the inputs. There are clearly mechanisms of the input-output process that we have yet to understand, but these numbers show that outputs are large and contribute substantially to downstream reaches.

Biodiversity

In British Columbia more than 75% of terrestrial vertebrates are closely associated with riparian areas (Morgan and Lashmar 1993). Headwater riparian areas may be unique in that they typically have a closed canopy with some different structural features than those found along larger streams with broad floodplains or openings in the canopy. Among the species found primarily in small streams are our 2 stream amphibians, the Pacific giant salamander (*Dicamptodon tenebrosus*) and the tailed frog (*Ascaphus truei*). Both these species are most commonly found in streams smaller than about 5 m bankfull width within British Columbia (Richardson and Neill 1995, 1998). Another vertebrate group found primarily along small streams are the water shrews.

The invertebrate faunas of streams and riparian areas are poorly known for any area of British Columbia, but we know enough to be able to point to groups of species that are uniquely associated with headwaters. There are several families of aquatic insects that are primarily associates of small, steep streams, including the mountain midges (Deuterophlebiidae) and net-winged midges (Blephariceridae). Two caddisflies of the genus *Eocosmoecus* have only been reported from about 6 sites each in British Columbia, and barely a handful of sites elsewhere in the Pacific Northwest (Wiggins and Richardson 1989).

The smallest of stream channels tend to have intermittent flow through late summer in British Columbia (unless spring-fed) and yet support a limited fauna of aquatic invertebrates. We do not yet know if there are species unique to these intermittent headwaters, but it is clear that some species reach their highest densities there and that they are by no means biological deserts (Muchow and Richardson 2000).

Cumulative Effects

Small changes in the biological and physical environment of headwaters may be incremental, leading to cumulative changes downstream. The magnitudes and directions of some of these changes are currently under study (Shrimpton et al. 2000, J. Richardson and P. Kiffney unpubl. data). It is clear that the effects of land use change the rates, timing, and variances of various processes, so it is not a simple matter to add and scale-up small changes to predict consequences downstream.

DISCUSSION

Small headwater streams play an important but unquantified role in the functioning of watersheds. The effects of changes in headwaters as a result of land use may be small and cumulative, but there have been few studies that have addressed any of the biological links between parts of a watershed at this scale. The uncertainty about management needs of headwater ecosystems and their role in maintaining healthy downstream environments is in part a consequence of the focus on fish habitat as the primary goal of riparian protection in western North America. While other objectives for riparian management have been articulated, the effectiveness of current guidelines have yet to be evaluated for goals other than protecting fish habitat. Some of the effects of not considering headwater protection include changes to subsidies of organic matter to downstream reaches, loss of a unique habitat, and small, incremental changes to downstream habitats.

Organic Matter

Stream ecosystems are intimately tied to the landscape through which they flow, and the streamside, or riparian,

habitat is critical to the conservation of streams. The many influences of riparian areas on streams include supply of structural materials (large woody debris), inputs of organic matter, and shading. Over the past 2 decades the importance of organic matter inputs from riparian vegetation has been emphasized by many authors (Hynes 1975, Cummins et al. 1989). The 2 primary sources of biologically available energy in streams are primary production (algae and vascular plants) and detritus (organic materials from vegetation surrounding the stream). In many small, forested streams more than 80% of the energy base of the food web comes from organic inputs. Recent studies have demonstrated the critical role of organic matter input to maintenance of the productive capacity of streams (Richardson 1991, Wallace et al. 1997).

Given that small streams receive the highest loading of organic matter of any channels in a watershed, and given the high number of small channels, this represents a large supply of energy to downstream reaches. Although our calculation that the export of organic matter to downstream reaches from headwaters exceeds 4 times the inputs cannot be correct, it underlines 2 basic facts. First, we still do not understand all aspects of the dynamics of organic matter in small streams. And second, clearly there is an enormous energy subsidy to downstream environments from the inputs of riparian vegetation in headwaters.

Biodiversity

Many organisms are dependent upon headwater streams and their riparian habitat. I have used aquatic insects and stream amphibians as examples, but there are many other species dependent upon the maintenance of aquatic and adjacent riparian habitat for the persistence of their populations. In 1 of the only studies published to date on Pacific Northwest intermittent streams, many species of aquatic and semi-aquatic insects were found in these Oregon headwaters and nowhere else in the watershed, including many new species (Dieterich 1992, Dieterich and Anderson 1995). Many of these species are probably not rare or at risk; it's just that we have put very little collective effort into looking at these habitats.

In British Columbia, estimates show that the highest biodiversity of vertebrates and vascular plants is associated with riparian areas, more so than with any other component of the surrounding landscape (Morgan and Lashmar 1993). In British Columbia, about 75% of the species listed as being vulnerable to human activities are riparian zone associates. Many of the birds and bats that frequent riparian areas do so because of the supply of adult aquatic insects near the stream. Consideration of these organisms depends on a broader perspective on watershed management and riparian protection.

Buffer strips have been shown to be effective in preserving the biodiversity and productivity of streams in California (Newbold et al. 1980). In that study, streams lacking reserve strips had fewer species and high densities only of opportunistic species, to the detriment of the rest of the fauna, another aspect to biological diversity. There have been few studies of the need for riparian vegetation by the adult stages of aquatic insects, but some observations suggest that insects gather around tall trees for mating aggregations, and there is no indication of what effects removal of trees may have. Loss of headwater habitats may create problems for dispersal and eventual recovery of populations restricted to small streams and their riparian areas. We do not yet know the requirements for the conservation of many of these organisms at any spatial scale.

Cumulative Effects

A consideration that we are only just beginning to address is the problem of cumulative effects. While small streams may, by themselves, be a common and small habitat, it is likely that the downstream effects of disturbing large numbers of small channels add up. These cumulative effects are likely to be manifest as changes to process rates, as well as their temporal and spatial variation. These effects may include loss or altered timing of organic matter inputs, alteration of temperature regimes, changes in peak-flow hydrology, and alteration of sediment supply and routing. Lack of riparian vegetation around these small streams is reported to result in warmer temperatures downstream, higher rates of sediment transport, and rapid depletion of organic matter by export (Hartman and Scrivener 1990). These effects may be magnified downstream by the degree to which a watershed has been converted from natural vegetation. We have yet to design appropriate studies on the scale necessary to detect such cumulative effects against the large natural variation in downstream environments.

Many critical elements of the stream and streamside ecosystem may be maintained by the provision of buffer strips of riparian forest. However, these reserves are only applied to larger streams with fish populations, to the possible neglect of headwater channels. The estimates for buffer widths are obtained by measuring the sources of woody debris in streams and their point of origin (Murphy and Koski 1989), and have resulted in a generally applied buffer strip width of 30 m. Whether these reserve strips will fulfil any or all of the objectives of stream and riparian conservation is uncertain and the subject of several ongoing studies. One such study is currently underway at UBC's Malcolm Knapp Research Forest to test the effectiveness of current and alternative riparian reserve guidelines for the maintenance of stream and riparian ecosystems (J. Richardson unpubl. data). The collective wisdom of fisheries biologists and stream ecologists supports the protection of streamside areas as an important aspect of protection for fish stocks, water quality, and other biodiversity objectives. Nevertheless, guidelines for riparian protection remain largely untested. Testing of current guidelines and development of creative

alternatives to those guidelines are required before we can offer the flexibility in riparian management desired by both industry and conservation interests.

MANAGEMENT IMPLICATIONS

There are many questions about the role of headwater channels in maintaining the function of watersheds and providing services for downstream environments. There may be adequate protection for small streams in protected areas, but there is a high degree of uncertainty. Beyond harvesting, there are other forest practices that may pose risks for headwater streams and may propagate downstream, for instance application of brush control (mechanical or chemical) and requirements for green-up in adjacent blocks. The seral stage distribution of riparian forest within a watershed may result in sufficient numbers of streams in the right spatial configuration to conserve the headwater stream ecosystem, but we have yet to even frame the appropriate question to test this. It will be unlikely that it is necessary to put reserves around every small stream, but we should make sure we know what the consequences are of not doing so. Watershed restoration and other conservation measures can only work if we take a watershed perspective. The high degree of uncertainty demands research into the role of these small channels in terms of cumulative effects, habitat loss, effects on dispersal, and downstream effects

ACKNOWLEDGEMENTS

This research was supported by Forest Renewal British Columbia and the University of British Columbia. I appreciate the assistance of many people with the work that underlies the ideas presented here.

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Uncertainties Linking Tailed Frog Habitat and Population Dynamics With Riparian Management

Glenn D. Sutherland
Centre for Applied Conservation Biology, University of British Columbia
3004 - 2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada
gsland@interchange.ubc.ca

John S. Richardson
British Columbia Ministry of Environment, Lands and Parks
and Department of Forest Sciences
University of British Columbia
3004 - 2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada

Fred L. Bunnell
Centre for Applied Conservation Biology
University of British Columbia
3004 - 2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

Efforts to quantify risks of alternative forest and habitat management options to populations of species considered vulnerable to forestry practices are usually limited by uncertainties in their habitat associations, as well as basic life history and population trend data. Identifying "disturbance/management envelopes" within which populations are likely to persist remains difficult for several reasons: species' requirements are incompletely known; only some life stages are practically accessible for monitoring; and different environmental factors may operate on populations across a species' geographic range. We confront this problem for the tailed frog (*Ascaphus truei*) in British Columbia by combining 2 approaches. First, we use hierarchical classifications of habitat variables measured in province-wide surveys to identify contingencies in environmental conditions that may alter likely responses to habitat protection measures. Topographic, climatic, and geological features are related to occurrence patterns of this species, while additional attributes of streams contribute to patterns of observed abundance in some areas. Second, we use simple population and disturbance models to explore potential effects of different disturbance/management scenarios on long-term persistence of populations of this species. Demographic factors associated with growth and survivorship interact with habitat factors to substantially alter long-term risks resulting from management. Fundamental uncertainties about factors altering growth and survival of instream life stages, and the long-term effects of forestry practices on survivorship, and fecundity attributes of older age classes encumbers sustainable conservation decisions at present.

Key words: *Ascaphus truei*, demography, disturbance regimes, headwater streams, riparian, risk analysis, tailed frog.

Efforts to quantify risks of alternative forest and habitat management options to populations of species considered vulnerable to forestry practices are usually limited by uncertainties in their habitat associations, as well as in basic life history and population trend data. This problem is especially acute for montane, stream-dwelling, riparian species, often living in non-fish-bearing headwaters that are difficult to access and inventory. We chose the tailed frog (*Ascaphus truei*) as a case study species for examining implications of such uncertainties on habitat management for 3 reasons: 1) potential impacts on harvest due to conservation measures for the species could have enormous impact on allowable harvests, because of the density of headwater streams in some areas; 2) its primary habitat (small headwater streams) is important to other poorly known species, so some results may be transferable to them (Sutherland and Bunnell in press); and 3) the species is Blue-listed throughout its range in British Columbia.

Putative effects of forestry on this species are difficult to determine on the basis of either extensive survey results, or from more intensive, replicated comparative studies. Current studies are showing that growth rates may be increased for larval stages following harvest, with possible decreased time to metamorphosis (Richardson and Neill 1998). Some researchers report decreased larval densities in second-growth stands, potentially reflecting long-term cumulative impacts of altered streambed structure, or decreased algal productivity beneath dense second-growth forests (Richardson and Neill 1998). Yet any observed responses of

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp

tailed frogs appear modified by local characteristics of the creek and site.

In this study, our objectives were to: 1) examine relationships between macro- and micro-scale environmental factors and current distribution and abundance of tailed frogs; 2) examine sensitivity of tailed frog populations to alternative hypotheses about the frequencies and effects of habitat disturbances; and 3) determine how sensitive different populations might be to potential effects of forestry practices. We used 2 approaches in this study: 1) an exploratory, hierarchical classification procedure of habitat variables measured at the regional, watershed, and reach level in relation to occurrence and abundance data to determine ecological relationships broadly related to the species' distribution and abundance; and 2) site-level population and disturbance models to investigate plausible bounds on unknown parameter values that may offer insight into long-term responses to management.

METHODS

Habitat Analysis

We first assessed existing habitat data for interpretable relationships that may apply broadly across the tailed frog's geographic range in British Columbia. To do this, we used inventory data for tailed frog presence/absence (and in some cases density) collated by Dupuis and Bunnell (1997), and added large-scale climatic, geological, and topographic measures, and estimates of area harvested since 1978 at different elevations (Sutherland and Bunnell in press). After using partial regression methods to filter out spatial autocorrelation among variables (Legendre 1993), we used a Classification and Regression Tree (CART) algorithm (Brieman et al. 1984) to split the data into conditional subsets that best explain the variation in the response variable (e.g., presence/absence or density) under study. Although simple interpretation of CART models is problematic when sampling is incomplete or biased (e.g., our samples were taken from access roads, restricting possible inferences about effects in undeveloped sub-basins), the models can be used as parsimonious descriptions of the data, and as an informative step in developing an ecological understanding of the dataset.

Site-level Population Modeling

To determine long-term risks to populations' responses to disturbances and forest practices, we constructed female-only, stochastic, stage-based population models in RAMAS GIS (geographic information system; Akcakaya 1997). Using these models, we examined plausible ranges of stochastic variation in habitat productivity, as well as expected changes in habitat quality as a result of forestry practices and disturbances for different types of populations and environmental conditions that may be found across the species' range in British Columbia. For this analysis we did not model dispersal between adjacent populations, because dispersal patterns in tailed frogs are very poorly known and other studies suggest that dispersal ranges are short (Daugherty and Sheldon 1982, Bunnell et al. 1998).

Our models divide the tailed frog life cycle into 5 stages: eggs and hatchlings, small tadpoles (<2.0 cm snout-vent length [SVL]), and large tadpoles (>2.0 cm SVL) living in the stream; and pre-reproductive metamorphs and reproductive adults living in riparian and nonriparian habitats. To create stage-based projection matrices, we needed to estimate (by stage) the probability of surviving and growing into the next stage (G_i), the probability of surviving and remaining in the same stage (P_i), and fecundity (F_i). Because we know little about the variability of growth and survival rates among individuals, we assumed that each individual within a stage is subject to the same probabilities of survival and stage duration. Published reports suggest tadpoles may take 1–4 years to reach metamorphosis, depending on geographic location and features of the natal stream. This same kind of variation may be present in other life stages.

We based our disturbance scenarios on 2 sizes of headwater stream: a small tributary, 500 m in length, representing a small area occupied by a tailed frog population; and a larger, 2-km stream. We defined 2 population types on the basis of growth conditions: a high-productivity population, with age at first reproduction of 5 years, females breeding every year, and a realized rate of increase of 0.29; compared with a low-productivity population with age at first reproduction of 8 years, 40% of females breeding every other year, and a realized rate of increase of 0.01. For both population types, average survivorship for each stage was 0.4, 0.5, 0.5, 0.6, and 0.7 respectively (values estimated from literature sources and from Duellman and Trueb [1984]).

We varied 3 types of environmental effects on the modelled populations. Annual changes in growing season length, temperature, and precipitation altered stage-specific vital rates (growth, survivorship, and fecundity). Infrequent catastrophic disturbances to streambeds and adjacent riparian habitat (e.g., debris flows, landslides, flooding) reduced abundance in affected stages during the year of disturbance. Putative long-term effects of forest harvest via sedimentation into streams, riparian habitat loss, and reduced light penetration through mid-seral, second-growth stands affected survivorship and habitat carrying capacity of the riparian zone. The particular scenarios we used are described in Table 1. Each scenario was simulated 100 times, with a simulation length of 200 years.

Table 1. Deterministic states and stochastic events defined for the site-level population model.

Model component	Parameter value	Ecological interpretation
Deterministic states		
population "type"[a]	low or high	local habitat productivity
habitat area	5 ha or 20 ha	stream length x riparian width
density dependence	none or strong	resource limitation on population
Stochastic events		
catastrophic mortality	1:20 yrs or 1:100 yrs	landslides, debris flows, floods
annual fluctuations in survivorship, growth and fecundity	10% coefficient of variation (CV) or 40% CV	variation in snowpack, temperature, and precipitation
Management actions		
removal of riparian vegetation	none or regeneration in 10 yrs	potential survival and growth effects on instream larvae and metamorphs
forest/riparian succession effects	none or 50% reduction in carrying capacity for 30 yrs	long-term effects on population through altered habitat productivity

[a] See text for definition of the growth and survivorship values for population types.

RESULTS AND DISCUSSION

Macro-scale Relations between Habitat, Distribution, and Abundance

The resulting classification trees for predicting occurrence patterns in tailed frogs are shown in Figure 1, with summary patterns in explained variation for occurrence and abundance variables shown in Table 2. Our analyses suggest several key findings. First, environmental variables operating at larger spatial scales (macro and meso scales, as defined in Table 2) account for more of the fit of the occurrence models than do micro-scale characteristics of the streams and

Table 2. Deviance explained by the CART models for tailed frog occurrence and abundance (see text) among the various classes of predictor variable. Scales defined by range of detectable spatial autocorrelation in the original variables (macro = 80–120 km; meso = 25–60 km; micro = <25 km). Source: Sutherland and Bunnell (in press).

Response class	Predictor class	Scale	Coast	Kootenays
Occurrence	Precipitation[a]	macro	13.7	14.1
	Location and extent of recent harvest	meso		26.1
	Bedrock lithology	meso	19.2	1.3
	Topography[b]	meso	12.4	1.3
	Stream substrate[c]	micro	3.7	23.3
	Stream temperature	micro		
	Stream width	micro		
	Fine organic debris	micro		–
	Reach slope	micro		16.5
Abundance[d]	Precipitation[a]	macro	46.9	(not estimated)
	Location and extent of recent harvest	meso		
	Bedrock lithology	meso	12.7	
	Topography[b]	meso	–	
	Stream substrate[c]	micro		
	Stream temperature	micro	9.6	
	Stream width	micro	0.5	
	Fine organic debris	micro		
	Reach slope	micro	0.5	

[a] annual precipitation (mean), summer precipitation (mean), both measured in mm.
[b] elevation, aspect, stream gradient.
[c] % cover of boulders, cobbles, and fines.
[d] For coastal sites, abundance estimates are calculated as absolute density (tadpoles/m^2), and models fit only for sites where frogs were present.

adjacent forest cover (Table 2), although stream substrate and reach slope are important classification variables in the Kootenays (Table 2; Fig. 1b). Variables operating at a larger scale also account for more variation in density in the coastal dataset (Table 2), although micro-scale stream characteristics (particularly substrate composition) are important in distinguishing sites with high tadpole density from those with low tadpole density. Second, at the site level, the CART models reveal how interactions between primary topographic and geological factors may influence responses of tailed frogs at the site level. Our regional comparison reveals that, while there was overlap in classes of habitat variables predicting presence/absence in the 2 regions (Table 2), the relative importance of each candidate variable differed between the 2 regions. In particular, variables linked directly or indirectly to forest harvesting (areal extent and location of recent harvest and stream substrate) were more important in the Kootenays than in coastal sites. Such differences in the importance of macro-scale environmental variables relative to micro-scale variables may in part reflect differences in regional hydrology regimes (Alila 1997), forest types, and histories of forest management in the 2 regions. They may also reflect artifacts of the restricted spatial distribution in occurrence records in the Kootenays (Dupuis and Bunnell 1997) limiting the ability of any statistical technique to adequately separate components of variation. Finally, variables directly reflecting forest management practices adjacent to sample sites (e.g., presence or absence of riparian buffers) did not appear in any of our classification trees, and sub-basin effects of logging appeared only in the Kootenays occurrence tree. This suggests physical setting has a greater influence on occurrence and abundance of larval frogs than do adjacent forest practices. Effects of logging differ among streams partly as a function of stream size and gradient, and their influences on sedimentation and primary production; these effects need to be taken into account in planning.

We interpret these results as follows (see also Sutherland and Bunnell [in press] for more detailed discussion). Practices designed to conserve existing tailed frog tadpole populations (buffers around streams, avoiding unstable slopes, minimizing stream crossings) will have the greatest chance of success if: 1) they are situated in intrusive or metamorphic bedrock formations; 2) they are in areas of moderate annual rainfall, with relatively high summer precipitation; and 3) they are in watersheds with low or moderate previous levels of harvest. Conservation efforts are likely to be ineffective where weathering products from local geology produces fine-grained or rubbly stream sediments and in

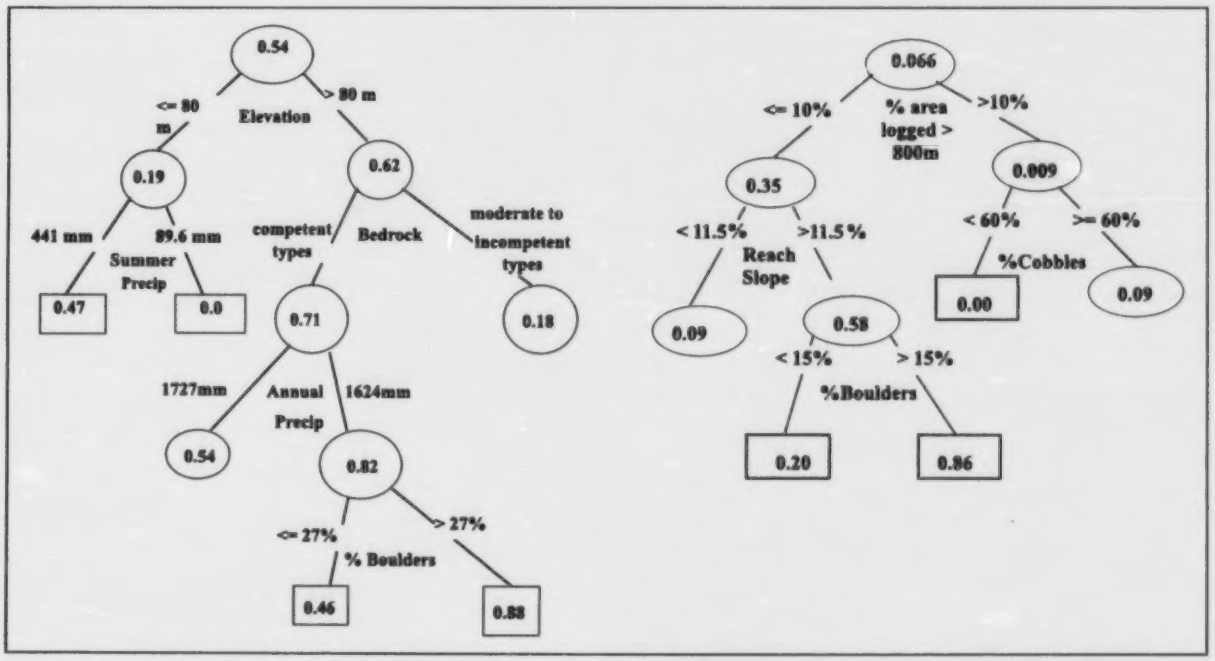

Figure 1. Classification and Regression Tree (CART) models for tailed frog occurrence in the western coastal mountains of British Columbia (left tree; n = 217 sites), and the eastern Kootenays (right tree; n = 136 sites). Numbers inside the circles (intermediate nodes) or rectangles (end nodes) are the proportion of sampled sites with tailed frogs present. The splitting predictor variable and its mean or threshold value is shown in each branch at each node. Source: Sutherland and Bunnell (in press).

sites with high probability of dry periods during the summer.

Tailed frogs in British Columbia may be resident as tadpoles in streams for up to 4 years (e.g., Wahbe 1996), suggesting particular vulnerability to frequent disturbance in sensitive areas, such as steep stream escarpments and areas of highly dissected terrain (Rollerson et al. 1998). Likewise, we know that tailed frog populations are remarkably discrete within streams, and show strong genetic differences among streams (Bunnell et al. 1998) suggesting low movement potential. Those observations emphasize the importance of maintaining on-site habitat quality in the most productive areas. We also suspect that fecundity of tailed frog adults is linked to temperature and productivity of riparian areas.

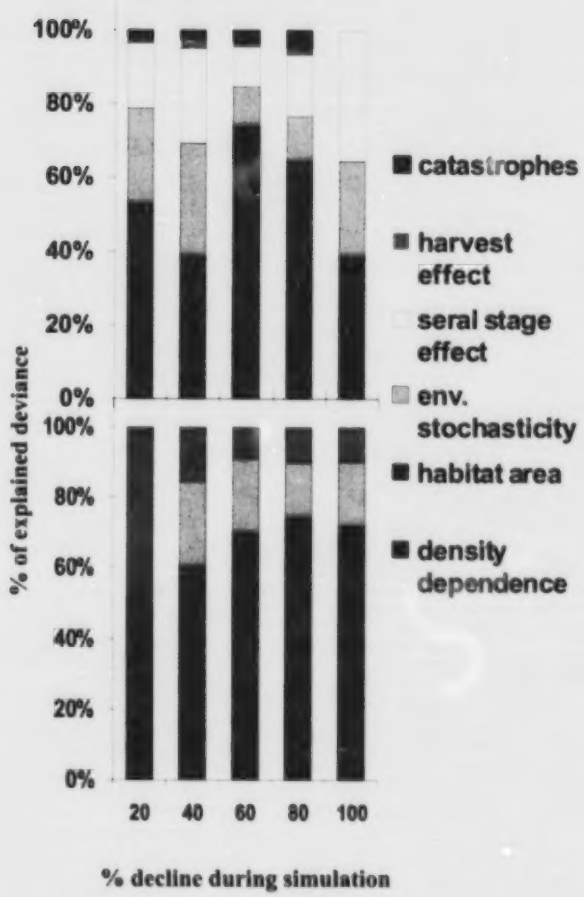

Figure 2. Percentage of the total deviance explained by different factors potentially contributing to a given % decline from initial population size for the "high" (top graph) and "low" (bottom graph) productivity population types.

Uncertainties in Population Responses

For each population matrix m, we examined the sensitivity of its finite rate of increase (λ_m) to proportional changes in 2 life cycle parameters: growth rate and survivorship in each life stage. Elasticities of λ_m with respect to F_i, P_i, and G_i (see Caswell [1989] for derivation) revealed that on average, λ_m was more sensitive to the probability of survival with growth (average elasticity for G_i = 0.545) than to probability of survival without growth (average elasticity for P_i = 0.319) or fecundity (average elasticity for F_i = 0.111). These results imply that forest management practices that directly affect rates of growth (via increased stream temperatures or increased food availability) following harvesting may have a greater effect on population persistence than on altered survivorship or adult fecundity.

Our primary indicator of model population vulnerability is defined as the risk of declining by a given amount from the initial population for each scenario. For example, across all

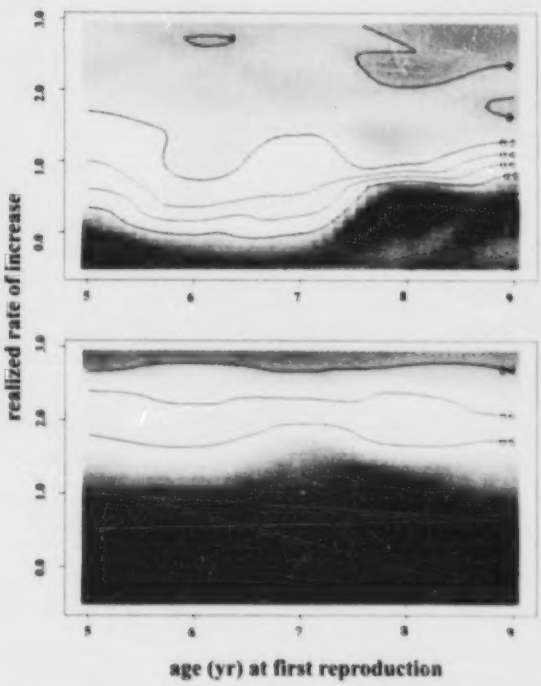

Figure 3. Contour diagrams of the probabilities of a population declining by 80% from the current state at least once during a simulation for a range of growth and survivorship rates. Probability values shown at the right of each contour line. Two contrasting scenarios are illustrated: no harvest or forest/riparian succession effects simulated (top graph), and both harvest and succession effects simulated (bottom graph). See text for other parameter settings.

our scenarios, our "low" productivity population type was much more likely to experience a decline of 80% from the initial population size at some point in the simulation (probability of 0.97), than was the "high" productivity population (probability of 0.59).

We examined resultant levels of population vulnerability in relation to the different values of the input parameters to understand the relative contribution of each simulated factor in the 2 types of populations. Because some factors are categorical, we again used a CART analysis to examine patterns of variation. In general, we found that factors representing long-term resource availability (e.g., density-dependence and small area of occupied habitat) are involved in causing declines in both population types, although density-dependence is more important in the "high" population type, and habitat area is more important in the "low" population type (Fig. 2). The immediate impacts of harvesting affect declines in the "low" population type, while the long-term effects of forest succession on carrying capacity of riparian habitat are important in the "high" population type (Fig. 2). Other sources of environmental variation contribute to declines in both population types, although catastrophic effects do not alter the already high likelihood of declines in the "low" population type. From these comparisons, we infer that many populations of the tailed frog are likely to be at higher risk from cumulative effects of habitat alteration (loss of habitat area, and gradual declines in habitat carrying capacity) than from short-term impacts of streambed disturbance, or the immediate impacts of forest harvest.

Given limited understanding of the ecological linkages between habitat alteration, survivorship, and growth rates, how sensitive could populations across a wide range of environmental conditions be to potential effects of forest practices on riparian and adjacent forest habitats? We examined this for small, density-dependent populations existing in habitats of low levels of environmental variation, and altered their growth and survivorship profiles to represent a range of habitat productivity. We compared their vulnerabilities (measured as probability of declining by 80% from initial population size) to effects of 2 scenarios: 1) no simulated harvest and succession of riparian and upland forest; and 2) simulations including both harvest and forest succession. In general, fast-growing populations appear relatively less vulnerable to natural environmental fluctuations and disturbances to habitat than do slow-growing populations (Fig. 3: top graph). Cumulative effects of forestry practices appear to have potential to place populations in virtually all environments at moderate to high risk of extirpation over the long-term (Fig. 3: bottom graph). While quantitative estimates of the relative vulnerability of populations to these scenarios depends on the details of how forest practices influence each life stage, the qualitative conclusion appears clear. As a result we caution that long-term persistence of tailed frog populations through harvest rotations is in doubt unless measures to maintain riparian habitat integrity and productivity are adopted for these small streams.

Taken together, the results of this study suggest that for this species, prescriptions for protection of their riparian habitats need to account for site-to-site variation in habitat productivity as well as the longer-term cumulative effects of landscape management on carrying capacity. Simulations suggest populations are at greater risk from habitat alterations altering growth and survival rates over a multiyear period than from year-to-year environmental variability, especially if they live in less productive sites. Habitat protection options for headwaters need to consider sustaining the stability and productivity of both streambeds and the riparian zone over long time periods to be effective.

ACKNOWLEDGEMENTS

We thank T. Wahbe, L. Dupuis, and P. Friele for field data and discussions. Research support was provided through grants from Forest Renewal British Columbia, the British Columbia Ministry of Environment, Lands and Parks, the British Columbia Ministry of Forests, and Wildlife Habitat Canada to JSR and FLB.

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Clearcut Timber Harvest and Movement Patterns in Tailed Frogs

Brent M. Matsuda
Department of Forest Sciences, University of British Columbia
2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada
bmatsuda@interchange.ubc.ca

John S. Richardson
Wildlife Branch, British Columbia Ministry of
Environment, Lands and Parks and
Department of Forest Sciences, University of British, Columbia
2424 Main Mall, Vancouver, BC, V6T 1Z4 Canada

ABSTRACT

Despite many studies of tailed frog (*Ascaphus truei*) larvae, there has been little study of juvenile and adult habitat use, patterns of movement, and dispersal. Grids of pitfall traps and drift fences were established in "control" sites (mature stands >81 years, n = 3) and clearcut habitats (<5 years, n = 3) to determine whether: 1) movement became limited to upstream/downstream activity only within the immediate riparian area in clearcuts without riparian buffers; 2) tailed frog abundance was lower in clearcut stands than in mature stands; and 3) pre-reproductive juveniles were the main dispersers. In clearcut sites, the highest proportion (44%) of frogs moved downstream, and the lowest proportion (11%) moved away from the creek. There was no notable difference between movement towards the creek (20%) and upstream (24%). Within mature sites, the highest proportion of frogs (40%) moved upstream, followed by movement towards the creek (32%). The lowest proportions moved away from the creek (13%) and downstream (15%). Captures were highest within 5 m of creekside in both clearcuts (41%) and mature sites (35%), followed by captures at 45-m (26% and 31% respectively) and 25-m intervals (22% and 23% respectively), with the lowest number of frogs caught at 65 m (11% in both habitats). Abundance of frogs was lower in clearcuts (n = 63) than in mature sites (n = 88). Of the frogs caught in clearcuts, 78% were juveniles and 22% were reproductive-age adults. Within mature stands, 53% of captures were juveniles and 47% were adults. Marking techniques indicated that frogs caught within the study sites were predominantly new individuals moving through the area. These findings suggest that although movement direction differs in clearcuts, distance travelled from streamside does not significantly change, with juveniles being the main dispersers in clearcuts.

Key words: *Ascaphus truei*, clearcut, dispersal, forestry, mature stands, riparian, tailed frog.

Numerous studies have found amphibian populations to be negatively correlated with clearcutting (see deMaynadier and Hunter 1995 for review). In Canada, the tailed frog (*Ascaphus truei*), a provincially Blue-listed (vulnerable) species in British Columbia, has only been studied in the larval stage, with little information known about habitat use, movement, and dispersal after metamorphosis. This lack of information has been largely due to the difficulty of studying the species owing to its small size (<15 g) and elusiveness, being located in relatively inaccessible mountain streams, and with both tadpoles and adults hidden under rocks and logs during daylight hours (Nussbaum et al. 1983). This information is critical to addressing issues of habitat management, gene flow, and metapopulation dynamics. Without knowledge of movements within and between habitats, preservation of 1 habitat type (e.g., riparian habitats only) may not be sufficient if adjacent upland habitat is needed for population maintenance but left unprotected.

The objective of this study was to evaluate whether clearcut harvesting influences tailed frog movement relative to that which occurs in mature stands. The general hypotheses and predictions being tested are:

H_1: Movement patterns become limited to upstream/downstream movement within riparian zones in clearcut (altered) areas, in contrast to patterns of habitat use in mature stands.

H_2: Clearcutting will result in lower tailed frog abundance than in mature stands.

H_3: Pre-reproductive juveniles will be the main dispersers.

STUDY AREA

Three replicates each of clearcut habitats (<5 yr old) and mature second-growth sites (>81 yr) were established in the Chilliwack Valley of southwestern British Columbia to assess differences in movement patterns between the 2 habitat types. This area lies within the Coastal Western Hemlock biogeoclimatic zone of British Columbia (Meidinger and Pojar 1991), within the Cascade Range, approximately 120 km east of Vancouver. Study sites were selected based on feasibility of

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

installing pitfall stations, alongside S5 or S6 streams[1] where preliminary surveys revealed tailed frog larvae.

METHODS

Sixteen pitfall stations were established in each study site, spaced 20 m apart in a 4 X 4 grid pattern (Fig. 1a). Each station consisted of clusters of 4 pitfall traps quartered by 4 guiding "drift" fences arranged in an "X" pattern such that direction of movement could be determined (upstream, downstream, towards or away from the stream). Stations closest to the streams were within 5 m of streamside. Pitfall traps were fashioned from 40-cm PVC pipes vertically dug into the ground with 25-cm-high plastic drift fences extending 5 m out from the centre of the trap station (Fig. 1b). A total of 384 pitfall traps and drift fences were used for the 6 study sites.

Traps were checked daily during a 3-day/3-night trapping period from May to September 1998, and were closed between trapping sessions. Increased precipitation during October and November allowed traps to be continuously left open and monitored approximately every 4 days. Traps were left open for an average of 50 nights at each site.

All frogs caught in pitfalls or while walking through the sites were weighed, measured, sexed, aged, and marked by injection of a coloured, nontoxic elastomer dye under the skin of the hind leg, and individually marked by toe-clipping. Pitfall frogs were released in the quadrant opposite the one where they were caught (the assumption being that this was the intended direction), and frogs encountered during "walk-throughs" were released at point of capture.

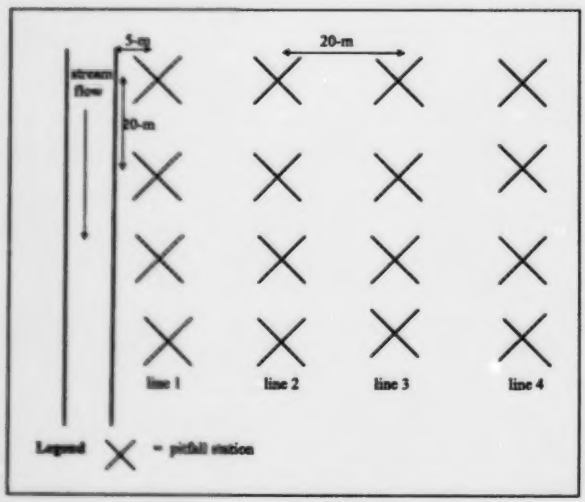

Figure 1a. Arrangement of pitfall trap stations in each study site.

[1] Streams outside of community watersheds, which are not fish streams: S5 streams are ≥3 m wide; S6 streams are <3 m wide (Stevens et al. 1995).

RESULTS

Pitfall traps were effective for sampling *Ascaphus* in both habitat types, with capture rates highest during the spring and fall months (Fig. 2). Captures generally decreased after June, coincident with the hot, dry period, then increased sharply in October, coincident with increased precipitation, and peaked in November. Between June and November, 88 frogs were caught in the mature stands and 63 in the clearcut sites. Two of the mature sites were not established in May, and exclusion of the 17 captures for this month in clearcuts indicates a substantially lower overall abundance in clearcuts. Breakdown by age class revealed that within mature stands 53% of frogs caught were pre-reproductive juveniles and 47% were adults, whereas in clearcuts 78% of frogs caught were juveniles and 22% were adults. Less than 10% of frogs were recaptured throughout the entire study.

Analysis of pitfall data indicated that within both habitat types, the lowest proportion of tailed frogs moved away from the streams (13% in mature sites, 11% in clearcuts; Fig. 3). Within mature stands, the highest proportion of frogs moved upstream (40%), in contrast to clearcuts, where the greatest proportion of frogs moved downstream (44%). In mature sites, 32% of captures were moving towards the stream and 15% were moving downstream. In clearcuts, 24% of captures were moving upstream and 20% were moving towards the stream.

The highest proportion of frogs was caught within 5 m of streamside in mature sites (35%) and clearcuts (41%), followed by captures at 45-m (31% and 26% respectively) and 25-m intervals (23% and 22% respectively), with the lowest number of frogs caught at 65 m (11% in both habitats; Fig. 4). In general, the percentage of frogs captured decreased as distance from streamside increased.

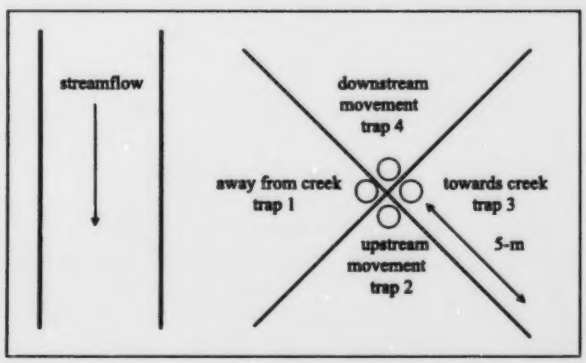

Figure 1b. Arrangement of pitfall traps at each pitfall station.

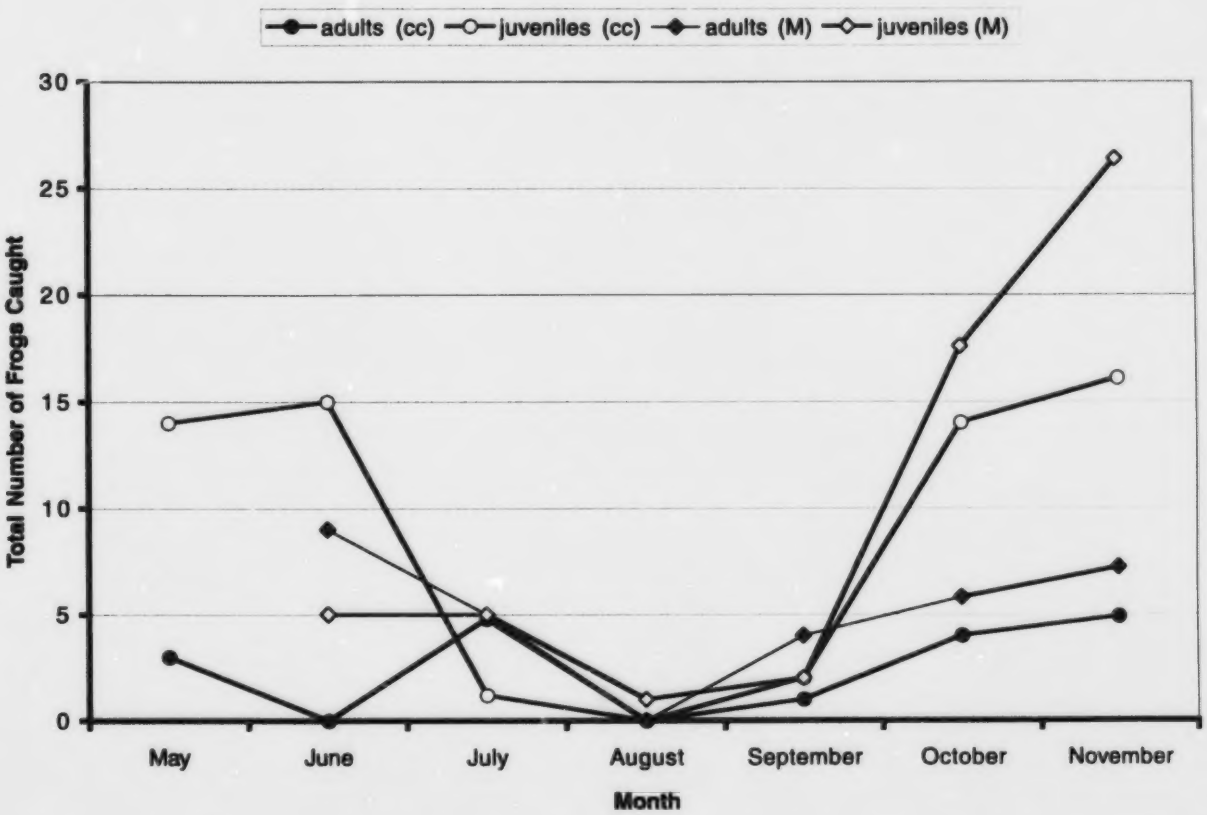

Figure 2. Monthly variation in tailed frog capture rates amongst age classes, and between clearcuts (CC) and mature stands (M).

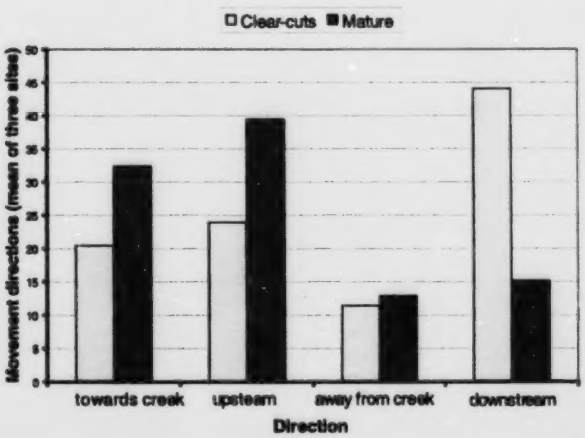

Figure 3. Mean proportion of tailed frogs moving in each direction.

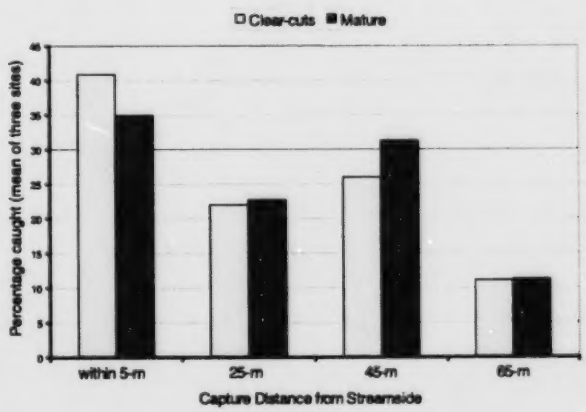

Figure 4. Mean proportion of tailed frogs caught relative to distance from streamside.

DISCUSSION

These results suggest that, regardless of habitat, tailed frogs prefer to move within the immediate riparian zone (<5 m from streamside) upstream and downstream. Our findings indicated that frogs in mature stands preferred upstream movement, supporting the findings of Metter (1964), who postulated that *Ascaphus* may move upstream to headwater creeks for summer refuge. Landreth and Ferguson (1967) found frogs moved downstream to larger creeks, possibly to facilitate breeding in more stable waterways rather than small headwaters prone to drying up. In this study this may have been the mechanism behind the large downstream movement in clearcuts. The clearcut site with the greatest overall number of *Ascaphus* had a large S3/S4 stream[2] within 25 m of the bottom of the trap grid, and a steep escarpment to climb within 500 m of the top of the grid. The low proportion of frogs captured moving away from streamside may suggest that tailed frogs are conservative in their movement efforts away from known sources of moisture (i.e., streams/riparian zones). This is supported by the high proportion of frogs caught within 5 m of the stream in both habitats. *Ascaphus* is the only North American frog specialized for and requiring cold, clear, fast-flowing mountain streams, and is physiologically more prone to desiccation than most other anurans (Daugherty and Sheldon 1982*a*, Nussbaum et al. 1983).

Based on a 63–73% recapture rate, Daugherty and Sheldon (1982*b*) found pre-reproductive *Ascaphus* to move greater distances than adults, which remained highly philopatric within 25 m of their original capture location. Recapture rates in this study were <10%, suggesting that frogs moving through the site were new individuals, or perhaps that trap shyness was occurring. Although distance travelled from streamside did not vary between clearcuts and mature sites, the high percentage of juveniles caught in our clearcut sites suggests that habitat alteration may favour juvenile dispersal, or perhaps indicates low survivorship to adulthood. Daugherty and Sheldon (1982*b*) found that movement activity did not vary with season, although our results indicated that frog activity was greatest during the spring and fall months.

MANAGEMENT IMPLICATIONS

This study indicates that, regardless of habitat alteration, *Ascaphus* tend to remain within the riparian area immediately adjacent to streamside. Within clearcut sites, movement largely occurred downstream and upstream, emphasizing the importance of stream habitats as movement corridors. Although survival to metamorphosis seems to be unimpeded in clearcuts, further studies are needed to assess juvenile rates of survival to breeding age, as our findings indicate that adult survivorship may be affected by clearcutting.

[2] Streams or portions of streams within community watersheds, which are also fish streams: S3 streams are 1.5–5 m wide; S4 streams are <1.5 m wide (Stevens et al. 1995).

ACKNOWLEDGEMENTS

We thank M. Bissonnette, M. Tressel, E. Kovics, R. Scott, M. Grant, D. Andrushko, A. Blaylock, and numerous volunteers for field assistance. K. Mallory, L. Frid, J. Curtis, G. Fried, L. Stad, K. Maxcy, M. Kim, R. Stott, S. Glenn, M. Porter, W. Neill, and C. Krebs provided additional assistance and advice. V. Lemay provided statistical and design advice. This project was funded through a Forestry Renewal British Columbia grant to J. Richardson.

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Defining Wildlife Habitat Areas for Tailed Frogs

Tanya R. Wahbe
Centre for Applied Conservation Biology
Forest Sciences Centre, University of British Columbia
3004-2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada
wahbe@interchange.ubc.ca

Fred L. Bunnell
Centre for Applied Conservation Biology,
Forest Sciences Centre, University of British Columbia
3004-2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada

R. Bruce Bury
Biological Resources Division, United States Geological Service
3080 Clearwater Drive, Corvallis, OR 97333, USA

ABSTRACT

The tailed frog (*Ascaphus truei*) is designated "at risk" or "of special concern" in British Columbia, Washington, Oregon, and California, yet none of these jurisdictions provide riparian buffers around small, non-fish-bearing streams where tailed frogs are found. We are investigating the movement patterns and dispersal abilities of tailed frogs in south coastal British Columbia. Results of larval research illustrate that larvae in streams flowing through old-growth forests moved about 7.5 times as far as larvae in streams flowing through clearcuts. Pitfall trapping results reveal that, although most frogs were found in streamside arrays, some individuals moved as far as 100 m from the nearest stream. Little is known about the dispersal abilities of tailed frogs, despite the implications to forest management. Our results will provide information on the potential benefits of streamside buffers and wildlife habitat areas to connectivity in fragmented landscapes.

Key words: *Ascaphus truei*, dispersal, headwater streams, mark–recapture, movement, tailed frog, wildlife habitat areas.

Tailed frogs live in small, fast-flowing streams with clear, cold water (Green and Campbell 1992, Leonard et al. 1993, Corkran and Thoms 1996), that often do not contain fish. Egg deposition sites are rarely found (Corkran and Thoms 1996), and when they have been found they were located close to the source of the stream (Brown 1975, Adams 1993), suggesting that movement of larvae is downstream. The species requires an unusually long time to develop, spending 1–5 years in larval form (Metter 1964, Metter 1967, Daugherty and Sheldon 1982*a*, Brown 1990, Wahbe 1996). Several studies have documented negative impacts of forest practices on tailed frog adults and larvae, including complete absence from clearcut areas (Metter 1968, Bury 1988, Bury and Corn 1988, Welsh 1990, Bury et al. 1991, Walls et al. 1992, Dupuis and Friele 1995). Some authors (e.g., Welsh 1990) have argued that recolonization of logged sites is critical to sustaining productive amphibian populations. Daugherty and Sheldon (1982*b*) found that 50% of reproductively mature *Ascaphus* remained in the same 20-m area of their previous capture, and concluded adult *Ascaphus* exhibit extreme site fidelity.

Headwater streams are classified as S4 (may contain fish), S5 (no fish), and S6 (no fish) in the *Riparian Management Area Guidebook* of the British Columbia Forest Practices Code (B.C. Ministry of Forests and Ministry of Environment, Lands and Parks 1995). These are Class C streams as described in the British Columbia *Coastal Fisheries/Forestry Guidelines* (B.C. Ministry of Forests et al. 1993). Headwater streams are afforded no mandatory protection under current laws unless the integrity of fish-bearing streams is at risk.

Although the tailed frog is designated "at risk" or "of special concern" in Oregon, Washington, and British Columbia, none of these jurisdictions provide riparian buffers around small, non-fish-bearing streams (Bunnell et al. 1997). In each case, forest practices legislation tends to be enacted as blanket prescriptions. Currently, British Columbia has the means to apply such prescriptions through the *Managing Identified Wildlife Guidebook* of the Forest Practices Code (B.C. Ministry of Environment, Lands and Parks and Ministry of Forests 1999). Through this research, we will determine the movement patterns and dispersal abilities of larval and metamorphosed tailed frogs in old-growth forests and clearcuts. Our goal is to provide scientifically credible guidance for the establishment of wildlife habitat areas for in-stream biological diversity, with emphasis on tailed frogs. Given both the documented (e.g., Bunnell and Dupuis 1995) and undocumented richness of small streams, many other species undoubtedly would benefit.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

STUDY AREA

Research was conducted near Squamish in southwestern British Columbia (Fig. 1), within coastal western hemlock (*Tsuga heterophylla*) forest. Sites were distributed within 4 drainages: the Squamish, Elaho, Ashlu, and Mamquam rivers, all tributaries to Howe Sound. Six headwater streams were selected, consisting of 2 treatments (n = 3): old-growth forest (>250 yr old) and clearcuts (5–10 yr old). Elevation of sites ranged from 500 to 1,000 m.

All streams in the Squamish Valley were selected after verifying presence of larvae (15-min searches), and reaches were chosen >50 m upstream from logging roads (except for Mamquam and Squamish clearcut sites, which were selected >50 m downstream from logging roads). Sites selected contained low-order streams without known resident game or anadromous fish.

METHODS

To determine the movement patterns of tailed frog larvae, area-constrained stream surveys were employed (Fig. 2; modified from Bury and Corn 1991, Shaffer et al. 1994) in the summers of 1994, 1995, and 1996. During these field seasons 80-year-old forest sites were also surveyed (n = 3). The dorsal fin of the tail was marked with V-shaped notches following Turner (1960) and Donnelly et al. (1994). Marks were "reach-specific" (e.g., 2 notches for larvae in reach 2). Area-constrained stream surveys and tailfin notches will also be used in the 1999 field season. Mark-recapture of larvae will be completed in 1999, using a modified stream sampling design. The design previously employed (Fig. 2; Wahbe 1996) will be modified in the field to include searches in larger stream sections (e.g., continuous 50 m).

Mark-recapture of metamorphosed tailed frogs using pitfall-trap/drift-fence arrays and visual implant fluorescent elastomers began in 1998 and will continue in 1999 and 2000. The construction and installation of all traps and fences was completed during the 1998 field season. Streamside and upslope arrays of pitfall traps with drift fences (Fig. 3) were modified from suggestions by B. Bury and L. Jones. In each of the

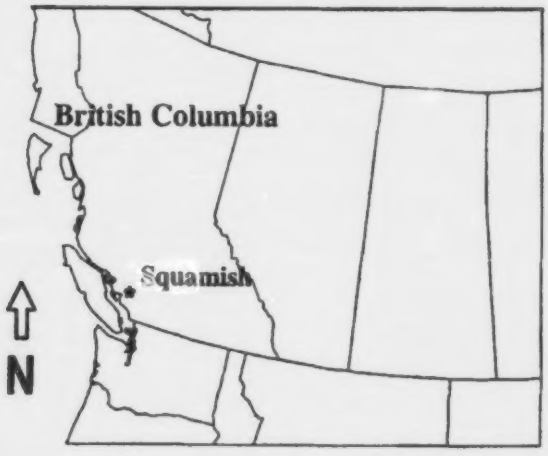

Figure 1. Map of British Columbia showing location of Squamish Valley.

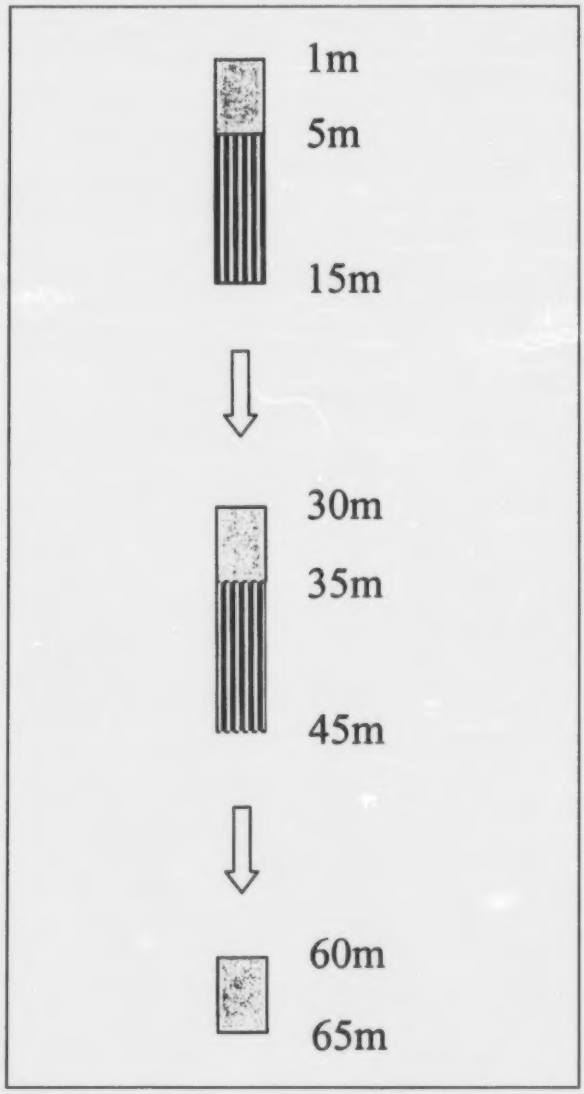

Figure 2. Stream sampling design. Initial marking occurs within 3 5-m reaches (solid boxes) separated by 25 m; sampling for recaptures also includes 2 additional 10-m reaches. Arrows indicate direction of stream flow.

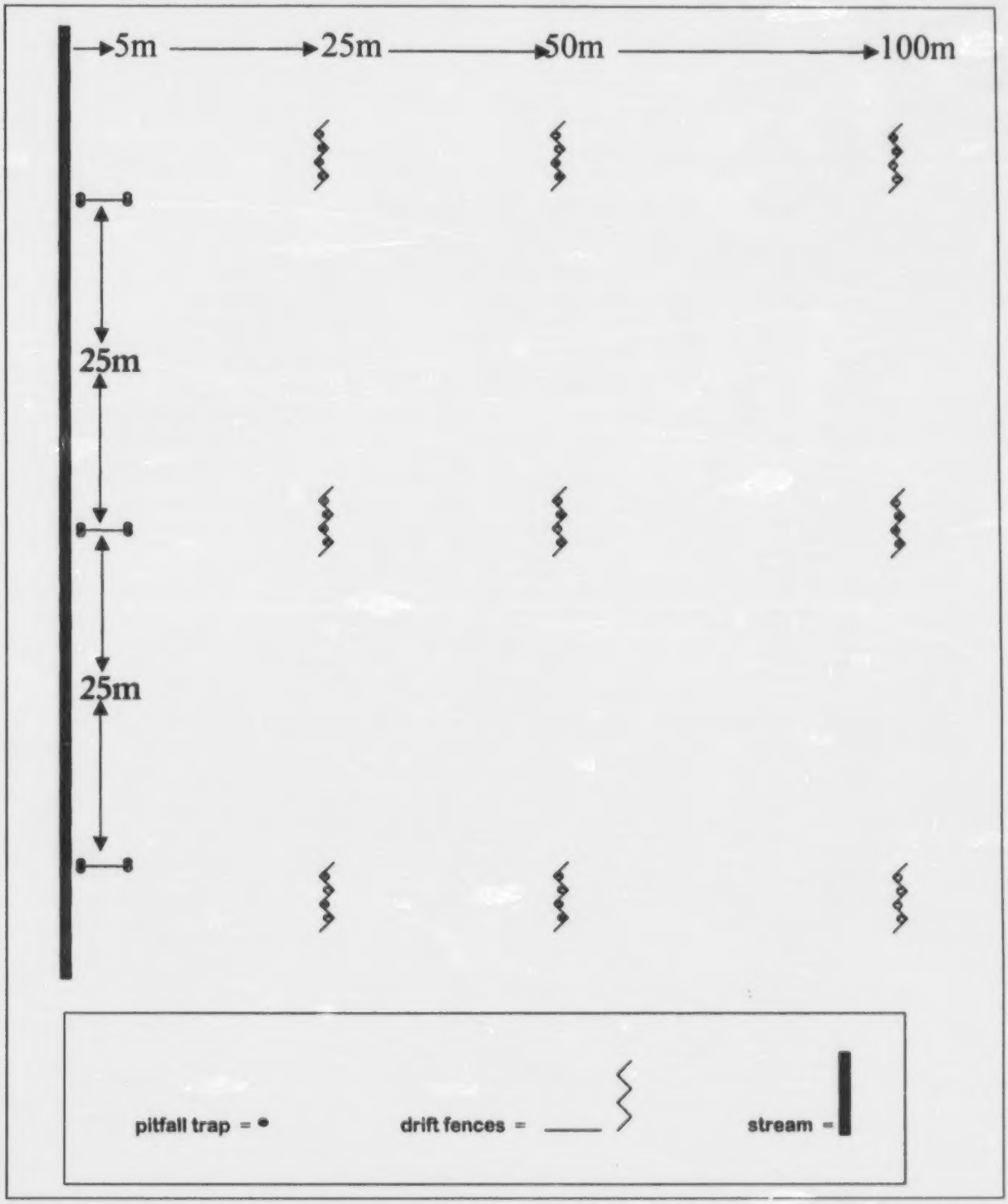

Figure 3. Streamside and upslope trap design (not to scale).

6 sites, 2 types of arrays (streamside and upslope) were constructed and installed (Fig. 3). There are 3 streamside arrays per site, each consisting of 4 pitfall traps and 1 drift fence. There are 9 upslope arrays per site, each consisting of 4 pitfall traps and 5 drift fences. Each of the 6 sites contains 48 5-m drift fences and 48 pitfall traps. There are a total of 240 m of drift fence and 288 pitfall traps in all 6 sites combined. Small mammal escape ropes were constructed and installed to reduce small mammal mortality (modified from Wind 1996).

All tailed frogs (as well as other amphibian species captured) were injected subcutaneously with visible implant fluorescent elastomers. Marks were unique to individuals. An animal care certificate and permit were obtained (University of British Columbia Protocol #A97-0035, British Columbia Environment Permit #C068233).

RESULTS

Larvae in streams flowing through old-growth sites moved on average 7.5 times further than larvae in streams flowing through clearcuts (Fig. 4; Wahbe and Bunnell in prog.). The maximum distance moved in old-growth sites was 64 m, while in clearcuts it was 3 m. The 1999 sampling period will allow final analyses of larval movement data.

Thirty-four tailed frogs were trapped over 21 nights of sampling (6,048 trap-nights). Sixty-eight percent of tailed frogs were trapped in old-growth sites (Fig. 5), where the rate of capture was twice as high as in clearcut sites (Fig. 6). Although most frogs were trapped in streamside arrays, some individuals moved as far as 100 m from the nearest stream. The individuals trapped at 100 m were newly metamorphosed tailed frogs moving away from the natal stream.

Of all tailed frogs trapped, 47% were reproductively mature adults and 38% were metamorphs (Fig. 7). Of the frogs moving parallel to streams, 88% were moving upstream (Fig. 8). Of the frogs moving perpendicular to streams (through upland forest), 56% were moving away from streams. With pitfall-trap/drift-fence array installations complete, mark-recapture of frogs will continue in 1999 and 2000.

DISCUSSION

Tailed frog larval movement hypotheses we are exploring include: 1) the stream channel barrier hypothesis, where log-jams in streams flowing through clearcuts create barriers to tadpole movement; 2) the algal productivity hypothesis, where streams with high algal biomass show low movement of tadpoles; and 3) the gradient/productivity hypothesis, where tadpoles in high gradient streams move less due to high productivity.

Given the uncommonly long larval stage of tailed frogs, movements by larvae may be particularly important to recolonization. After clearcutting, however, small streams often contain logging debris that could impede movement and recolonization by larvae from upstream egg deposition sites. Results to date show that larval dispersal abilities are significantly greater in streams flowing through old-growth forests than in streams flowing through clearcuts (Wahbe and Bunnell in prog.). All clearcut streams used in this study contained abundant logging debris, lending support to hypothesis

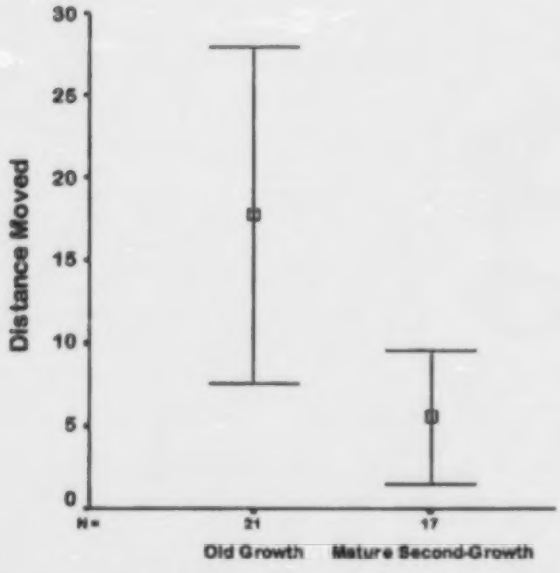

Figure 4. Average distance moved by tadpoles in each forest habitat type (1994, 1995).

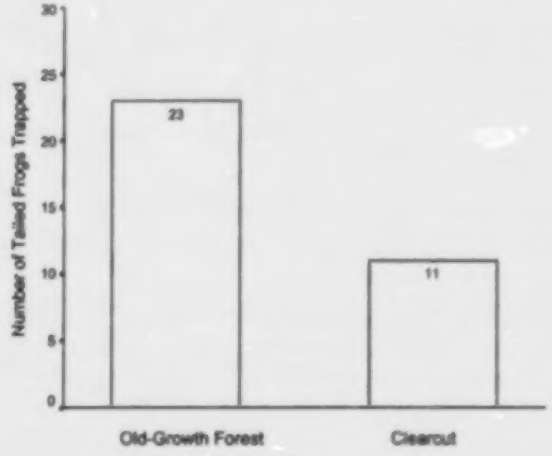

Figure 5. Total number of tailed frogs trapped in old-growth forests and clearcuts.

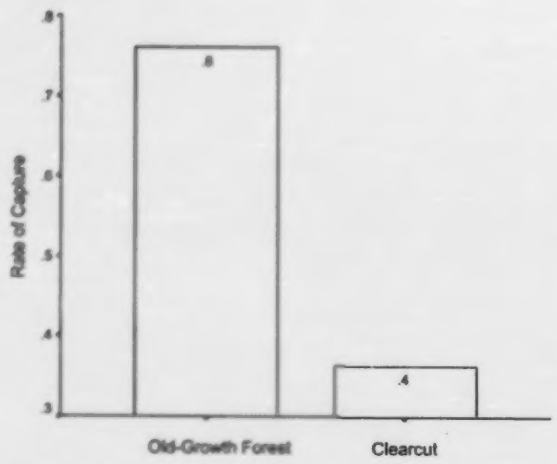

Figure 6. Captures per 100 trap-nights in old-growth forests and clearcuts.

1. Our data also support hypothesis 3, where high-gradient streams showed low movement of larvae. Hypothesis 2 will be tested during the 1999 field season.

Tailed frog post-metamorphic movement hypotheses we are exploring include: 1) the extreme philopatry hypothesis, where frogs remain within a restricted range, moving short distances along streams in both old-growth forests and clearcuts; 2) the stream dependency hypothesis, where frogs in clearcuts are more restricted to the stream than frogs in old-growth forests; 3) the age-specific movement hypothesis, where metamorphs and juveniles are responsible for most of the upland movements, while adults remain relatively sedentary, moving primarily along the stream; and 4) the macroclimatic movement hypothesis, where movements of frogs in wet environments are less restricted than movements in dry environments. Hypothesis 4 will be tested through comparisons between tailed frog movements in British Columbia and tailed frog movements in southern Oregon.

Preliminary trapping results reveal that, although most frogs were found in streamside arrays, some individuals moved as far as 100 m from the nearest stream. This demonstrates that the dispersal abilities of tailed frogs are much greater than previously suggested (Daugherty and Sheldon 1982*b*), and rejects hypothesis 1. More importantly, the individuals at 100 m were newly metamorphosed tailed frogs, lending support to hypothesis 3, where early age-classes are responsible for upland movements. The majority of frogs found in streamside arrays were reproductively mature and were moving upstream, which supports suggestions that tailed frogs breed during the fall in upper reaches of streams. This also lends some support to hypothesis 3, where adults remain relatively sedentary, moving primarily along the stream.

MANAGEMENT IMPLICATIONS

Forest management decisions should consider the ecological integrity of sensitive habitats and species, and the connectivity of ecosystems to ensure continued dispersal and movement of ecologically sensitive species across the landscape. It is important to ensure the protection of streams because they provide tailed frogs with breeding sites, over-wintering sites, and potential dispersal corridors for gene flow between populations. It may be equally important to ensure

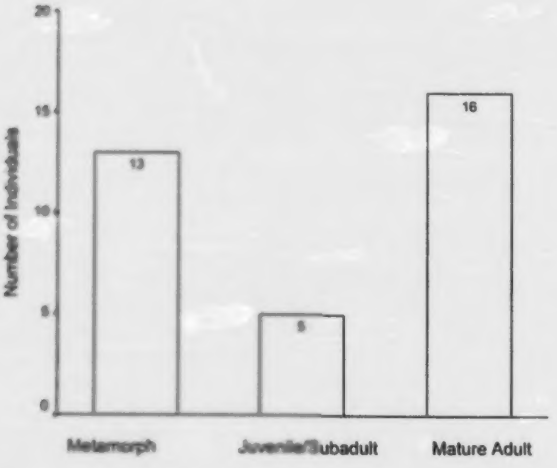

Figure 7. Number of tailed frogs trapped in each stage of developement.

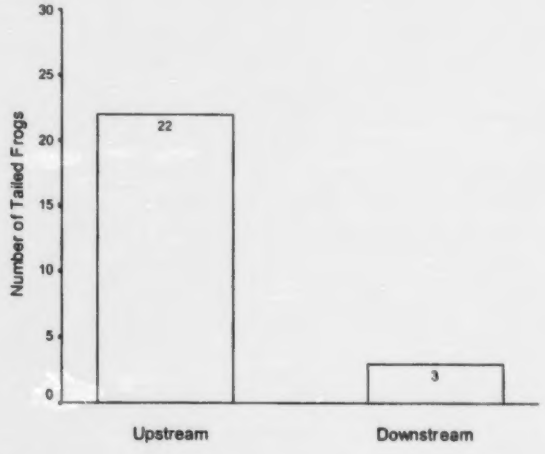

Figure 8. Number of tailed frogs moving upstream and downstream.

protection of the terrestrial habitat required by this species, and the value of riparian buffers needs evaluation.

Results of this research carry important conservation and management implications for tailed frogs and their habitats: cold, rocky streams, which are essential for many animals, including salmonid fishes. Our goal is to develop wildlife habitat areas or other alternative forest management designs to address the needs of identified wildlife.

ACKNOWLEDGEMENTS

We thank Wildlife Habitat Canada and Forest Renewal British Columbia for providing research funding. K. Sadowski, N. Job, L. Frid, K. Maxcy, N. Holling, S. Webber, J. Rodrigues, Y. Ota, and I. Houde assisted with fieldwork. Special thanks go to M. Wahbe, E. Jovel, R. Mooney, J. Dojillo, J. Arsenault, J. Peterson, B. Kreowski, and L. Burrows for volunteering their time on this project. T. R. Wahbe was supported by Forest Renewal British Columbia and a British Columbia Environmental Research Scholarship from the Ministry of Environment, Lands and Parks.

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Effects of Light and Nutrients on Grazer–Periphyton Interactions

Micheline A. Kim[1]
Department of Forest Sciences, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada

John S. Richardson
Wildlife Branch, British Columbia Environment, and Department of Forest Sciences, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

Forest disturbance can alter light and nutrient regimes in small streams, which in turn may influence grazer–periphyton interactions within the stream community. To investigate the influence of forest disturbance on grazer–periphyton interactions, we tested the effects of light, nutrients, and grazing by tadpoles of the tailed frog (*Ascaphus truei*) on periphyton standing crop and tadpole growth rates in 2 coastal streams. Periphyton biomass was found to be under simultaneous control by light and tadpole grazing. Light exerted a strong, positive effect on periphyton standing crop and tadpole relative growth rate, indicating a tight trophic coupling between grazers and their algal food resource. Tadpole growth rate decreased significantly with increasing tadpole density, showing that tadpoles are subject to intraspecific density-dependent interactions. The 2 streams differed in both abiotic limitation of periphyton biomass and tadpole growth. This study demonstrates that several factors may act together to control periphyton biomass and grazer growth rate, and that the relative importance of these factors may vary between nearby streams.

Key words: *Ascaphus truei*, density dependence, grazer, light, nutrients, periphyton, stream, tadpole.

Food webs are a central idea in ecology and represent graphically how nutrients and energy are cycled through species interactions within ecosystems (Wilbur 1997). Plants form the basis of most food webs and exert significant control over higher trophic level interactions (Power 1992). In stream ecosystems, primary production by periphyton plays a key role in determining the structure and function of the stream community. Stream periphyton communities are subject to both bottom-up forces, such as light, nutrients, substrate, flow, season, and disturbance, and top-down control by herbivory. The quantity and quality of the algal food resource influences the type and strength of interactions at higher trophic levels. At high densities, grazers can become food-limited and competition for a limited algal resource may result in intra- or interspecific density-dependent interactions (Hart 1987, Lamberti et al. 1987). Studies involving benthic invertebrates demonstrate that algal production can limit grazers in temperate streams through competition for food, reduced growth rate, and shifts in community composition of grazers (e.g., Lamberti and Resh 1983).

Small streams provide an opportunity to test the influence of abiotic factors from forest disturbance on in-stream community processes. Primary production by algae in small, forested streams is generally limited by a closed riparian canopy which may limit light reaching the streambed to <5% of full sunlight (Hill et al. 1995). Primary production may be further limited in oligotrophic streams by inherently low concentrations of both nitrogen and phosphorus. Removal of the riparian canopy due to forest harvest or windthrow can shift production from primarily allochthonous inputs to autochthonous production. How this shift in energy resources from forest disturbance impacts grazer–periphyton interactions in these systems is a complex, poorly understood issue.

Interactions of several factors may yield complex outcomes and therefore we designed a complete block, factorial experiment to assess the effect that changes in light and nutrients from riparian forest disturbance may have on grazer–periphyton interactions in small fishless streams. An amphibian grazer, the tailed frog tadpole (*Ascaphus truei*), was selected to study the link between lotic herbivory and primary production for 2 reasons. First, ecologically, they often occur at high densities in the predominantly fishless streams that they inhabit, are morphologically specialized for feeding on epilithic periphyton, and may be dominant herbivores in some streams (Lamberti et al. 1992). Second, from a conservation perspective, the tailed frog is currently Blue-listed in British Columbia and considered vulnerable to forest disturbance (Corn and Bury 1989). Changes in abiotic factors from forest disturbance may influence the occur-

[1] Present address: AXYS Environmental Consulting Ltd., P.O. Box 2219 - 1045 Mills Rd. West, Sidney, BC, V8L 3S8, Canada
akim@axys.net

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

rence, density, and biomass of tailed frog tadpoles by influencing the availability of the algal food supply. This study assesses the effects of forest disturbance on abiotic factors, periphyton biomass, and larval growth response, which may be a correlate of fitness for individual tadpoles. The major objectives of this study were to:

1. examine the influence of changes in abiotic factors (light and nutrients) on primary production by stream periphyton;
2. assess whether tadpole growth is food limited under different light and/or nutrient conditions; and
3. determine whether tadpole growth is influenced by density-dependent interactions.

STUDY AREA

The study was conducted in 2 high-gradient, permanently flowing, first-order creeks in the Chilliwack River drainage, approximately 115 km east of Vancouver, in southwestern British Columbia, Canada. The 2 creeks selected are approximately 6 km apart (straight-line distance). Both Klondike Creek and Dipper Creek were classified as S5 streams according to the *Riparian Area Management Guidebook* (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1995). Both streams flow through second-growth forest dominated by Douglas-fir (*Pseudotsuga menziesii)* and western hemlock (*Tsuga heterophylla*), with red alder (*Alnus rubra*) occurring within the riparian zone.

METHODS

From August to September 1997, light, nutrient levels, and tadpole density were experimentally altered to test for their effects on periphyton levels in experimental stream enclosures in 2 creeks. Two light levels (shaded and unshaded), 2 nutrient levels (ambient and high), and 4 tadpole densities (0, 29, 57, and 86 tadpoles/m^2) were manipulated in experimental stream enclosures in each stream in a 2 x 2 x 2 x 2 complete block, fully factorial design. Each treatment combination was replicated 4 times in each stream. Densities were determined by the random assignment of 0, 1, 2, or 3 tadpoles to enclosures within each array. Each array consisted of 8 enclosures, each measuring 0.035 m^2. Enclosure design was modified from Lamberti and Feminella (1996). Additional arrays with enclosures measuring 2 times (0.07 m^2 = medium) and 3 times (0.105 m^2 = large) the size of small enclosures were used to allow for 2 additional density treatments (10 and 14 tadpoles/m^2, respectively). Eight arrays of small enclosures, 4 arrays of medium enclosures, and 4 arrays of large enclosures were used at each stream.

Experimental enclosures were placed in uniform areas within runs approximately 5–15 cm below the surface of the water to simulate conditions where tadpoles were most likely encountered within the stream channel (personal observation). Unglazed ceramic tiles (5 x 10 cm) and stream cobbles were placed in each enclosure and conditioned for 5 weeks before introduction of tadpoles. A total of 152 *Ascaphus truei* tadpoles per stream were collected from within a 200 m reach at each stream. Individual tadpole total lengths and weights were recorded and tadpoles randomly assigned to enclosures. Light levels were manipulated using 90% industrial shade cloth (American Horticultural Supplies) to simulate closed canopy conditions. Surface irradiance was measured at several locations in both the open and shaded sections on 10 August 1997 using a PAR-quantum light sensor (LI-COR, Nebraska). Nutrient levels were manipulated using slow-release osmo-coated nutrient pellets deposited 10 m upstream of half of the enclosures. The remaining half of the enclosures in the upstream portion served as ambient nutrient controls.

During the 6-week experiment, the outside of each enclosure was cleaned of debris every 2 days and enclosures were checked to ensure tadpoles had not escaped. At the end of the experiment, tiles and cobbles were collected for analysis of periphyton biomass (ash-free dry weight, AFDW) and chlorophyll *a* concentration. Water chemistry and stream temperature were measured during the experiment. At the end of the 42-day experiment, 95% of tadpoles were recovered, weighed, measured, and released. Periphyton was removed from cobbles and tiles by scrubbing the entire surface with a toothbrush and rinsing with distilled water. To determine periphyton biomass, half the sample was filtered through a Whatman GF/F filter (Whatman Inc., Kent, U.K.), dried at 55°C for 24 hours, weighed, combusted at 550°C for 1 hour, and reweighed to estimate AFDW by loss on ignition. Chlorophyll *a* analysis followed methods outlined in Strickland and Parsons (1972). Water samples were placed directly on ice and frozen for nutrient analysis. Nutrient concentrations were measured using a Technicon Auto Analyzer II (Technicon Corporation).

Prior to statistical analysis, data were tested for normality. Tadpole relative growth rate (RGR) was calculated as: RGR (g) = $[(\text{weight}_{\text{initial}} - \text{weight}_{\text{final}})/\text{weight}_{\text{initial}}]$/no. of days of the experiment. Effects were evaluated with analysis of variance (ANOVA) and analysis of covariance (ANCOVA) using the PROC GLM function in SAS (SAS Institute 1996). Least squared means and standard errors generated from 1-way ANOVAs were plotted to depict tadpole growth response to density treatments.

RESULTS

Addition of nutrient pellets resulted in a 6-fold increase in nitrate concentration and a 2-fold increase in phosphate concentration in Dipper Creek. Shade cloth reduced light levels by 85–90%. Water temperature for Klondike Creek during the experiment ranged from 8.1 to 14.9°C (mean =

12.3°C). Water temperatures at Dipper Creek during the experiment ranged from 7.8 to 11.2°C (mean = 9.5°C). Size–frequency distributions for tadpoles at each creek indicate that tadpoles used in this experiment likely represent a single cohort of 1 -year olds. Slopes of allometric regressions differed by a factor of 1.5 between the 2 creeks (Dipper = 0.028; Klondike = 0.019), indicating that tadpole condition (e.g., weight/length ratio) before the experiment differs significantly between the 2 creeks with tadpoles at Dipper Creek weighing on average 50% more for a given length than tadpoles at Klondike Creek ($P < 0.0001$, $F_{1,300} = 160.2$).

Light and Nutrient Effects on Periphyton

ANOVA showed that light exerted a significant positive effect on chlorophyll *a* ($P < 0.0001$) and periphyton biomass ($P < 0.0302$). Light accounted for 44% of the observed variation in chlorophyll *a* abundance. However, light accounted for only 1% of the observed variation in periphyton biomass. Chlorophyll *a* abundance was 2–7 times higher under light conditions than in shade and showed a significantly greater response to light treatments at Dipper Creek than at Klondike Creek (light x stream interaction, $P < 0.0001$, Fig. 1). Periphyton biomass exhibited a smaller response to light treatments than did chlorophyll *a*. At Dipper Creek, periphyton biomass increased 30–40% in light but exhibited little response at Klondike Creek. Nutrients had no significant effect on chlorophyll *a* ($P = 0.802$) but had a significant positive effect on periphyton biomass ($P < 0.0001$). Overall, nutrients accounted for <1% of the variation in chlorophyll *a* abundance, and 28% of the variation in AFDW, resulting in a 25–50% increase in AFDW. Light and nutrients interacted significantly to influence both chlorophyll *a* abundance and periphyton biomass. Stream and light interacted significantly to influence chlorophyll *a* abundance; however, there was no significant interaction between stream and light for periphyton biomass.

Grazer Effects on Periphyton

Results of the ANCOVA show that tadpole density had a significant negative effect on chlorophyll *a* ($P < 0.0001$, $F_{1,121} = 11.14$) and on periphyton biomass ($P < 0.0001$, $F_{1,121} = 25.31$). The effects of grazing are most clearly demonstrated by contrasting the control (0 tadpoles) with the lowest density treatment (10 tadpoles/m^2, 1 tadpole) at Klondike Creek (Fig. 2). This result shows that the introduction of a single tadpole resulted in a 50% decline in chlorophyll *a* abundance relative to the no tadpole control. Differences between the 2 study streams significantly influenced periphyton biomass ($P < 0.0001$, $F_{1,121} = 42.6$), but had no significant effect on chlorophyll *a* ($P = 0.0796$, $F_{1,121} = 3.13$). There were no significant interactions between stream and density for chlorophyll *a* or periphyton biomass.

Grazer Growth Response

Tadpole relative growth rate decreased significantly with increasing tadpole density ($P < 0.0001$), which accounted for 39% of the variation in tadpole relative growth rate (Fig. 3). Differences between the 2 study streams accounted for 33% of the explained variation in tadpole relative growth rate ($P < 0.0001$). Tadpole relative growth rate was approximately 35% higher at Klondike Creek than at Dipper Creek across all density treatments. Light positively influenced tadpole relative growth rate ($P < 0.0001$) and accounted for 14% of the explained variation. Relative growth rate was signifi-

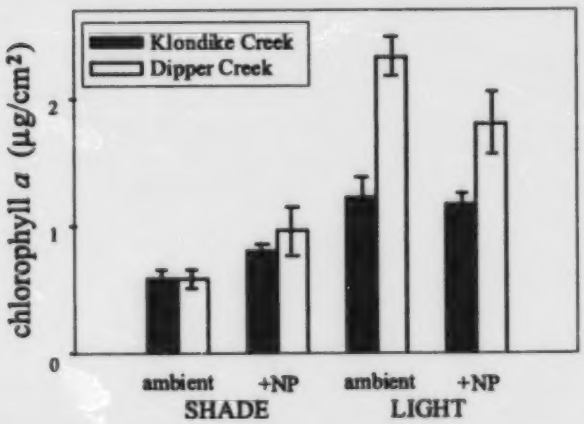

Figure 1. Effect of light and nutrients on chlorophyll *a*. Ambient treatment represents upstream control section; +NP represents downstream fertilized section. Bars represent means (±1 SE for all cobbles (n = 64/stream) from small enclosures.

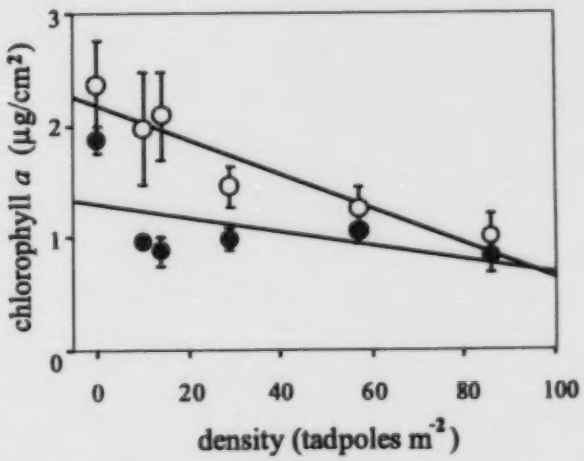

Figure 2. Relations between tadpole density and chlorophyll *a* for Klondike (●) and Dipper (○) Creeks. Dots represent means ±SE for all cobbles (n = 64/stream).

cantly higher in the unshaded treatments than in the shaded treatments at both Klondike (1.2x) and Dipper (1.5x) creeks. Nutrients had no significant effect on tadpole relative growth rate ($P = 0.491$). None of the interactions tested were statistically significant.

DISCUSSION

The results of this study indicate that changes in light and nutrient regimes in small, headwater streams can significantly change periphyton production, which in turn can influence tadpole growth rates. Enhanced growth rates of tadpoles may lead to shorter time to metamorphosis, larger size at metamorphosis, and/or enhanced survivorship of the juvenile stage (Duellman and Trueb 1994). The most striking result about tadpole growth was that differences between the 2 study streams were the most important factors determining tadpole growth, accounting for 33% of the explained variation in relative growth rate. Tadpole relative growth rate was 1.2–2.5 times higher at Klondike than at Dipper Creek. The most probable explanation for this difference is the warmer stream water temperature at Klondike Creek.

In this study, light was found to have a significant positive effect on tadpole relative growth rate at both streams. This growth is presumably mediated by an increase in primary production, and indicates a tight trophic coupling between tadpole grazing and the algal food resource. Nutrients had only a slight but nonsignificant positive effect on tadpole growth rate under unshaded conditions, which reflects the fact that nutrients had little effect on the periphyton food resource. The small, positive effect of nutrients under unshaded conditions indicates that light is the primary limiting factor, and that the positive effects of nutrients may only be expressed once periphyton production is released from light limitation.

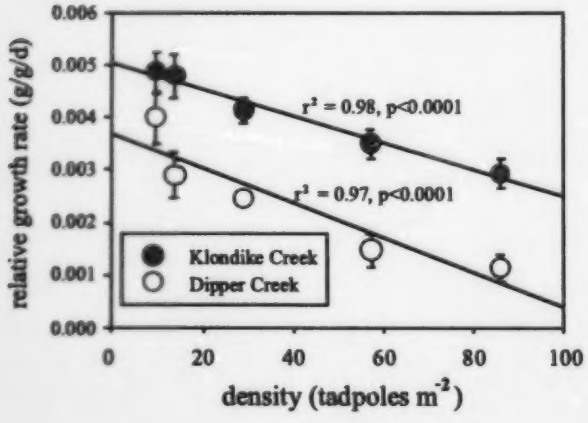

Figure 3. Effect of tadpole density on tadpole relative growth rate. Dots represent means (±1 SE for all tadpoles (n = 152/stream).

By applying a complete block, fully factorial design, this study demonstrates that the relative growth rate of *Ascaphus* tadpole decreases as tadpole density increases, indicating that tadpoles were subject to intraspecific density-dependent food limitation. Higher tadpole growth rates at Klondike Creek, despite the fact that food resources were lower, suggest that temperature differences between streams had a strong effect on growth rates. However, the fact that the growth response to increased larval density followed the same pattern at both streams is most logically explained by density-dependent food limitation. This conclusion is supported by decreases in tadpole growth rate and periphyton standing crop as tadpole density increases (Figs. 2 and 3). At Klondike Creek, where growth rates were higher, algae were depressed even at the lowest densities (e.g., 10 tadpoles/m^2), whereas Dipper Creek had a more gradual decline in algae with increasing tadpole density. If density dependent growth rates are linked to survival during the larval period, then survival rates will also show density-dependence, which can carry over to the juvenile and adult stages of the life cycle. Because population size for amphibians is largely determined by the larval stage, longer-term studies are needed to determine how changes in environmental conditions attributable to forest disturbance influence time to metamorphosis, size at metamorphosis, and survivorship of tadpoles and juveniles of *Ascaphus*.

Complex interactions require complex experiments to address causal mechanisms (Wilbur 1997). In this study, manipulation of abiotic and biotic factors to assess their relative importance on stream periphyton and grazer growth response resulted in complex outcomes. Light, nutrients, grazer density, and several interactions between these factors simultaneously influenced stream periphyton production and growth rates of tadpoles. Differences in the response of periphyton and tadpole growth to the treatments at the 2 streams demonstrate that the relative importance of these factors can be site specific. At Klondike Creek, bottom-up control by light was a more important limiting factor for periphyton standing crop than grazing. However, at Dipper Creek grazer control of periphyton was more important than light limitation.

MANAGEMENT IMPLICATIONS

This study demonstrates that several factors (light, nutrients, tadpole density) can act together to determine periphyton production and growth rates of tadpoles. Because growth rate may be a correlate of fitness for individual tadpoles, managers should consider how changes in environmental conditions within the riparian zone influence in-stream processes. Although increased insolation to the

stream may enhance primary production and grazer growth, other factors may reduce production, thereby making the overall effects of forest disturbance difficult to predict (Murphy and Hall 1981). The size and duration of these effects will vary between streams; therefore, a site-specific approach should be taken when managing for conservation of in-stream habitat. Finally, the tailed frog has a complex life history which includes a multiyear aquatic larval stage and a terrestrial juvenile and adult phase. Management of riparian forests for conservation of the tailed frog should consider habitat requirements for all life history phases.

ACKNOWLEDGEMENTS

We thank Dr. B. Neill and Dr. S. Hinch for their comments and support. Thanks to R. Stott for assistance in the field and D. White for assistance in the lab. Financial support for this project was provided by Forest Renewal British Columbia and the Donald McPhee Fellowship.

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Unexplored Diversity: Macroinvertebrates in Coastal British Columbia Headwater Streams

Christine L. Muchow
Department of Forest Sciences, University of British Columbia
2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada
cmuchow@interchange.ubc.ca

John S. Richardson
Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks, and
Department of Forest Sciences, University of British Columbia
2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

At the very tips of most drainage networks are small stream channels, many of which have no flow at times during the summer. These small, zero-order streams contribute a significant portion of cumulative stream length in a watershed, but are largely unexplored and receive little protection under current legislation. We studied the aquatic macroinvertebrate assemblage of 7 small coastal streams with a range of permanence of flow. Three streams have no detectable surface flow for periods in summer (i.e., intermittent flow). We hypothesize that the invertebrate assemblage in these zero-order streams does not completely overlap that of larger streams. Stream sites were sampled using a combination of emergence traps emptied every 14 days throughout the year. In even the smallest streams (<0.5 m bankfull width) with intermittent flow, true aquatic insects with 1-year life cycles were found emerging, even in periods when no flow was perceptible. Species richness in intermittent and continuous streams was approximately equal, while intermittent streams appeared to produce as much as twice the number of adult stoneflies as continuous streams. This study shows that even intermittent streams harbour a true aquatic fauna and, potentially, species for which little is known, and indicates that these unique habitats make an important contribution to their ecosystems.

Key words: aquatic invertebrate, coastal watershed, intermittent stream, Plecoptera, riparian.

The small streams at the tips of catchments in watersheds are variable in their periodicity of flow: some flow continuously throughout the year, and others only flow during precipitation events or when groundwater levels are high. At certain times in the year a portion of these small, zero-order streams have no perceptible surface flow; that is, they have intermittent flow. While zero-order streams constitute a significant proportion of stream length in a watershed (Sidle et al. in press), these systems have been largely ignored by scientific studies of aquatic fauna. Since the riparian zones in which these small streams exist contain valuable resources and habitat (Davies and Nelson 1994), their management and protection is related to their value as buffer strips, stream bank stabilizers, and fish and wildlife habitat (Mahoney and Erman 1984).

Despite the importance of riparian habitat to ecosystem function, and the concentration of biodiversity in aquatic environs, only 1 other study in the Pacific Northwest (Dieterich and Anderson 1995) has attempted to explore the benthic macroinvertebrate fauna of these unique headwater stream habitats. Threats to benthic invertebrates are important, as these organisms are key in the trophic chain of food webs in stream ecology and they support other organisms, such as salmonids, that are of cultural and economic concern in British Columbia (B.C. MOF and B.C. MELP 1995). Unfortunately, despite their unexplored status, these unique streams are afforded little, if any, protection under current legislation .

We know very little about biodiversity in headwater streams. As part of a larger, integrated project to test current riparian guidelines, we focused on aquatic insects in zero-order streams. We hypothesized that the invertebrate assemblage in these zero-order intermittent streams does not completely overlap, or is distinct from, that of larger and continuously flowing streams.

STUDY AREA

This study was located in the University of British Columbia's Malcolm Knapp Research Forest. This research forest is located in Maple Ridge, B.C., approximately 60 km east of Vancouver, in the rainy, temperate Fraser Valley foothills (elevation 140–450 m). The sites are located in the Coastal Western Hemlock biogeoclimatic zone, have glacial, primarily

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Table 1. Stream site parameters, University of British Columbia Malcolm Knapp Research Forest.

	Site B	Site F	Site I	Site G	Site H	East Cr.	Mike Cr.
Watershed area (ha)	13.5	12	12.6	68.9	40.8	44	25
Gradient (% slope)	14.25	12	20	3	3	14	12
Periodicity of flow[a]	I	I	I	C	C	C	C

[a] I = intermittent; C = continuous.

acidic soils, and are underlain by igneous rocks (Feller 1979). The 7 stream sites are located at the southwestern corner of the research forest, which experienced a large-scale fire in 1931 and abuts Golden Ears Provincial Park. All streams are designated S4, S5, or S6 under the Forest Practices Code of British Columbia (B.C. MOF and B.C. MELP 1995). They have a range of permanence of flow, with 3 having no detectable surface flow for periods in summer (i.e., intermittent flow). See Table 1 for experimental stream parameters.

METHODS

Stream sites were sampled using a combination of 2 emergence traps per site sampled biweekly throughout the year, and spring/fall Surber sampling. Aquatic macroinvertebrates were field preserved in propylene glycol (adults) and formalin (nymphs). Individuals were sorted and identified to species using Merritt and Cummings (1996) and other keys, and preserved in ethyl alcohol. For this paper, only emergent adult stoneflies will be considered as an index of the broader study.

RESULTS

In even the smallest streams (<0.5 m bankfull width) with intermittent flow, true aquatic insects with 1-year life cycles were found emerging, even in periods when no flow was perceptible. Some examples of these species include *Soyedina producta*, *Moselia infuscata*, *Despaxia augusta*, and *Zapada cinctipes*. All species found in intermittent streams were also present in continuous channels but not vice versa, indicating that the headwater fauna is mostly a subset of downstream fauna in these zero-order channels (Table 2). Plecopteran adults emerging have approximately the same species richness (Fig. 1), and higher densities (Fig. 2) in intermittent streams than in continuously flowing sites. In intermittent sites the number of individuals emerging approached twice that of continuous streams for the 2-week monitoring periods. *D. augusta* (Leuctridae) was often the largest contributor to intermittent stream biomass, where it reached its highest densities in all tabulated months.

Table 2. Stoneflies (Plecoptera) collected as adults from intermittent and continuously-flowing streams in the University of British Columbia Malcolm Knapp Research Forest.

Intermittent flow	Continuous flow
Alloperla perlosa	*Alloperla perlosa*
	Bolshecapnia sp.
	Chernokrilus sp.
Despaxia augusta	*Despaxia augusta*
Malenka spp.	*Malenka* spp.
Moselia infuscata	*Moselia infuscata*
	Nemoura sp.
Ostracerca fosteri	*Ostracerca foesteri*
	Osborus sp.
	Paraleuctra vershina
Pteronarcys californica	*Pteronarcys californica*
Soyedina producta	*Soyedina producta*
	Sweltsa sp.
Zapada cinctipes	*Zapada cinctipes*
Zapada oregonensis	*Zapada oregonensis*
	Visoka cataractae

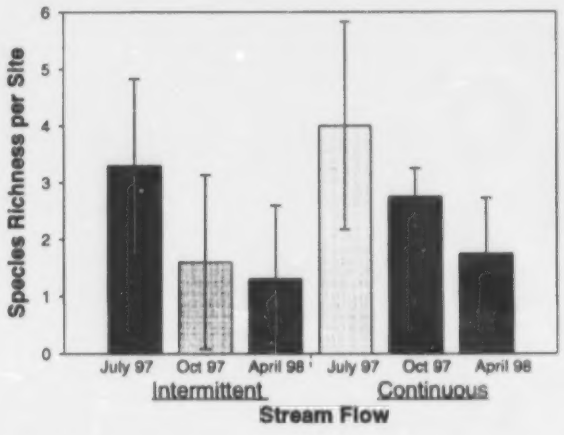

Figure 1. Stonefly (Plecoptera) species richness in intermittent and continuously-flowing streams at the University of British Columbia Malcolm Knapp Research Forest. Error bars indicate standard error for 3 intermittent and 4 continuously-flowing streams per sampling date.

DISCUSSION

Invertebrates are microhabitat-specific, and ongoing physical, biological, and geochemical conditions of the stream determine their abilities to persist (Sweeney 1984). Changes in microhabitat are reflected in changes in species composition and richness (Davies and Nelson 1994). Rather than being biologically barren, intermittent channels harbour a true aquatic fauna, emerging even in periods of no discernible flow, and rival the species richness of more stable continuous sites. *D. augusta*, with its 2-year life cycle, is able to complete its development in these periodically "dry" channels, and reaches its highest densities in intermittent streams. This suggests that suitable refugia exist for this species and others in the wetted sediments of these habitats, despite the periodic disappearance of detectable surface flow.

All species found in intermittent streams were also located in continuous channels, but not vice versa. This suggests that intermittent site fauna are a subset of continuous site fauna in these zero-order streams. Species missing from the intermittent assemblage appeared to be large-bodied, long-lived stoneflies (e.g., *Pteronarcys californica*) that required more oxygenated, faster flows than were available in the intermittent streams.

It is unknown at this point in our study whether the intermittent fauna is a unique subset of continuous and/or downstream fauna (higher order streams) and can be predicted from the periodicity of water flow by nested subset analysis. The relationship of the continuous and intermittent fauna to that of larger downstream reaches, (Richardson 1992), seems to indicate that downstream sites may act as sources for some species that may only survive in high water-flow conditions.

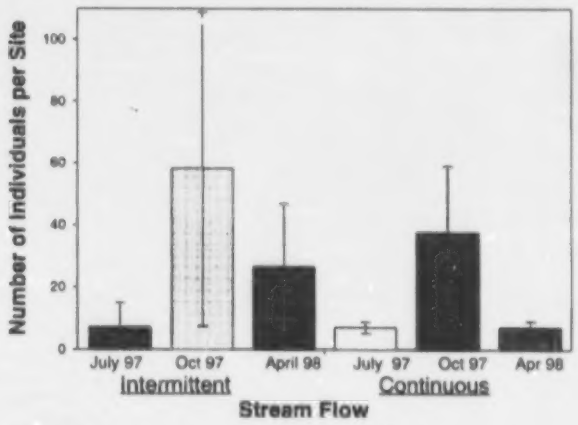

Figure 2. Stonefly (Plecoptera) adult density in intermittent and continuously-flowing streams at the University of British Columbia Malcolm Knapp Research Forest. Error bars indicate standard error for 3 intermittent and 4 continuously-flowing streams per sampling date.

MANAGEMENT IMPLICATIONS

Riparian zones play critical roles in maintenance of biodiversity (Clayoquot Sound Scientific Panel 1995), and in maintenance of aquatic biota (Mahoney and Erman 1984). Anthropogenic change in riparian zones, such as forestry, is of particular concern to benthic invertebrates, which are dependent on riparian vegetation for some stage of their life cycle and for maintenance of stream conditions for which they have evolved (Erman 1984, Mahoney and Erman 1984). Because all of these factors have the potential to critically impact the zero-order streams, it seems that further exploration is warranted, especially concerning the consequences of riparian management. These unique sites are not currently afforded protection, and consequently the species that exploit these zero-order streams may be detrimentally affected and their important contribution to these ecosystem's function lost.

ACKNOWLEDGEMENTS

We would like to thank J. Newhouse. This study was funded by a Forest Renewal British Columbia grant to J. Richardson.

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Effects of Stream Riparian Buffer Width on High Elevation Songbird Communities

Devon Haag
University of British Columbia, Forest Sciences
3rd Floor, Forest Sciences Centre
3041-2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada
devon@direct.com

ABSTRACT

Riparian systems provide critical habitat for wildlife species, particularly in drier climates such as those within the southern Interior of British Columbia, Canada. Forested riparian zones are also a source of valuable timber and are frequently subject to harvesting. Forest managers want to know how present harvesting practices can maintain the structural attributes required by different wildlife species. This research examines the capacity of the current riparian management guidelines to provide viable habitat for songbirds breeding in the Montane Spruce and Engelmann Spruce–Subalpine Fir biogeoclimatic zones of British Columbia. Specifically, the project assesses changes in avian communities with varying riparian buffer widths along small, high elevation streams in the Kamloops Forest Region. Sampling methods include songbird territory mapping, habitat mapping, and vegetation sampling. Research is ongoing; most data will be collected in spring and summer 1999. Preliminary results indicate peak species richness is found in medium width buffers (30 m) while several forest bird species occur only within wide buffers (60 m) and undisturbed forest. The management recommendations from this research will benefit those land managers responsible for maintaining biological diversity in montane forests.

Key words: behaviour, buffers, habitat structure, high elevation forests, Kamloops, neotropical migrants, riparian, songbirds.

Riparian zones contain valuable habitat (Bunnell and Dupuis 1993) and often receive disproportionately high utilization by wildlife. Water contributes to greater levels of plant productivity, creating a complex vegetation community that is attractive to many species (Thomas et al. 1979). In many areas of British Columbia, riparian zones support greater avian richness than occurs in all other habitats (Bunnell et al. 1995). Stream riparian ecosystems are also productive areas for tree growth and are frequently subject to timber extraction. Harvesting in riparian zones can reduce vegetation cover and wildlife habitat (Stevens et al. 1995). Habitat characteristics required by songbird species may be lost or compromised, affecting overall habitat quality and breeding success.

At the landscape level, harvesting in riparian and adjacent forests contributes to habitat loss and fragmentation. Forest fragmentation has been hypothesized as one of the principal causes in the decline of neotropical songbird populations in North America (Robinson 1992). The loss of habitat has negative consequences, including reduction of habitat area, greater edge-to-interior ratio, and decreased habitat quality (Wiens 1994). Area and isolation effects can result in lowered species diversity and fewer recolonization events following local extinctions (Freemark and Collins 1992, Ambuel and Temple 1993). The higher incidence of nest predation and brood parasitism experienced along fragment edges contributes to reduced nesting success (Gates and Gysel 1978, Wilcove 1985, Paton 1994).

The Forest Practices Code and the *Riparian Management Area Guidebook* regulate harvesting in watersheds throughout British Columbia (B.C. Ministry of Forests, and BC Environment 1995*a*). Streams are classified into 6 groups by average channel width, fish presence, and use as a community watershed. Riparian management areas (RMAs) must be retained along all streams in harvested sites. These areas are comprised of 2 sections: the reserve zone and the management zone. Harvesting is prohibited in the reserve zone and modified to meet specific objectives in the management zone. The size and management objectives of the RMA are specified by the stream class. All streams are assigned a management zone; streams possessing fish or watershed values are also assigned a reserve zone.

Proper management of riparian areas is critical to the conservation of ecosystem integrity and biological diversity. It has

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

yet to be determined if the riparian management guidelines are sufficient to meet the life requisites of different wildlife species. In particular, researchers are asking what RMA widths are required to maintain key ecosystem processes and habitat attributes necessary to support viable populations of species (SALASAN Associates and Dovetail Consulting 1996).

This project was proposed to assess changes in bird and vegetation communities along small, high elevation streams in response to varying riparian reserve (buffer) widths. The southern interior of British Columbia was selected as it represents a dry climate in which the significance of riparian habitats to songbird species may be heightened. High elevation streams were chosen because they are generally small, lack fisheries or community watershed value, and can be subject to high harvesting levels. The primary objectives are two-fold: (1) to determine the effects of decreasing buffer width on the structure and composition of high elevation breeding songbird communities, and (2) to determine what habitat attributes are required by forest songbird species and if these attributes are retained by riparian buffers.

STUDY AREA

The research is ongoing within Tree Farm Licence (TFL) 35, a 30,000 ha forest in the Kamloops Forest Region, approximately 25 km north of Kamloops, B.C. (Fig. 1). Weyerhaeuser Canada Ltd. is the licensee operating in TFL 35 and the surrounding areas. The project concentrates on the mid-elevation Montane Spruce (MS) and the high elevation Engelmann Spruce–Subalpine Fir (ESSF) biogeoclimatic zones. In the study area, these zones include the Thompson dry mild MS variant (MSdm2) and the Thompson dry cold ESSF variant (ESSFdc2).

Characterized by cold winters and relatively short summers, the MSdm2 contains climax stands of hybrid white spruce (*Picea engelmannii* x *glauca*) and subalpine fir (*Abies lasiocarpa*) (Meidinger and Pojar 1991). The understory is moss-dominated and includes falsebox (*Paxistima myrsinites*) and black twinberry (*Lonicera involucrata*) (Lloyd et al. 1990). The ESSFdc2, typified by long, cold winters and short, cool summers, is often dominated by Engelmann spruce (*Picea engelmanii*) with lodgepole pine (*Pinus contorta* var. *latifolia*) as a seral species. Understory layers are largely comprised of white-flowered rhododendron (*Rhododendron albiflorum*) and black huckleberry (*Vaccinium membranaceum*).

Five study sites were located in the MSdm2 and 5 in the ESSFdc2. Sites ranged in elevation from 1,100 to 1,550 m and were primarily characterized by benchy topography and an east–southeast aspect. Surrounding stands were composed of subalpine fir, spruce, and lodgepole pine with lesser amounts of Douglas-fir (*Pseudotsuga menziesii*). Average surrounding stand age ranged from 101 to 250 years.

METHODS

This retrospective study involved collecting only post-treatment data from existing harvested sites. In 1998, study sites were located throughout the TFL to encompass 2 replicates each of 5 treatments (stream buffer widths), including controls (Fig. 2). Blocking was by site, resulting in a randomized complete block design with 5 treatments, 2 blocks, and 10 sites in total. The 4 treatment widths were defined as: "very narrow" (x = 2 m), "narrow" (x = 13 m), "medium" (x = 32 m), and "wide" (x = 62 m). Controls were located along streams with >250 m of undisturbed forest extending into the upland. Site selection was limited to small streams (<5 m width) with recent harvesting (<4 yr) on 1 side and forested habitat on the other side.

Breeding Bird Surveys

Although a preliminary breeding survey was conducted in 1998, most data will be collected in the 1999 breeding season. Spot mapping (International Bird Census Committee 1970, Bibby et al. 1992) was selected as the most appropriate method for measuring territory density, territory distribution, and species–habitat associations.

One spot-map grid (4.5 ha) was constructed at each study site. Grids were 300 x 150 m, oriented lengthwise along the

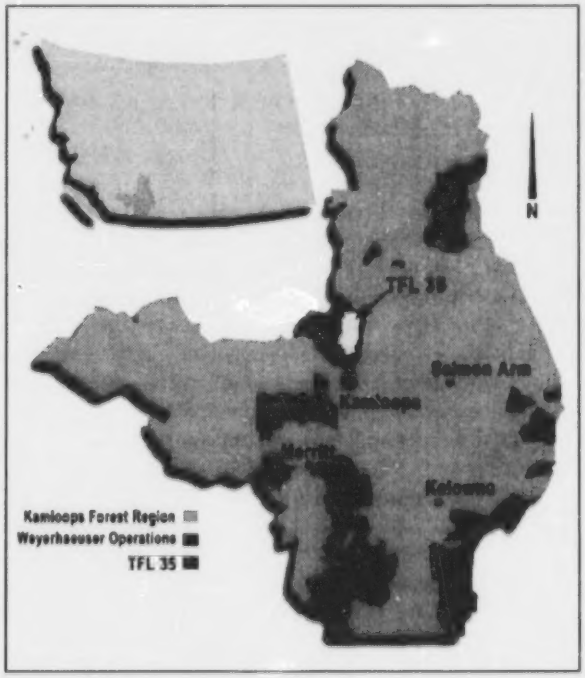

Figure 1. Location of Tree Farm Licence 35 within the Kamloops Forest Region.

stream and flagged at 25-m intervals. A Corvalis global positioning system (GPS) was used to record the locations of all flagged markers in every grid; points were plotted at a scale of 1:1,500 to produce census maps onto which all songbird observations were recorded. This approach facilitates the future entry of observations into a geographic information system (GIS) database, allowing the digital delineation of songbird territories and the analysis of inter- and intraspecific territory density and distribution.

Spot-map surveys were conducted from 30 June 1998 to 16 July 1998, with a single census performed at each site. The two personnel who conducted the surveys were trained, thus ensuring standardized observations. Censuses were completed between 0500 and 0900, and inclement weather was avoided. All bird observations and activities were recorded using the symbols given in Bibby et al. (1992). In the 1999 field season, surveys will begin by mid-May and continue through mid-July to guarantee the collection of sufficient data.

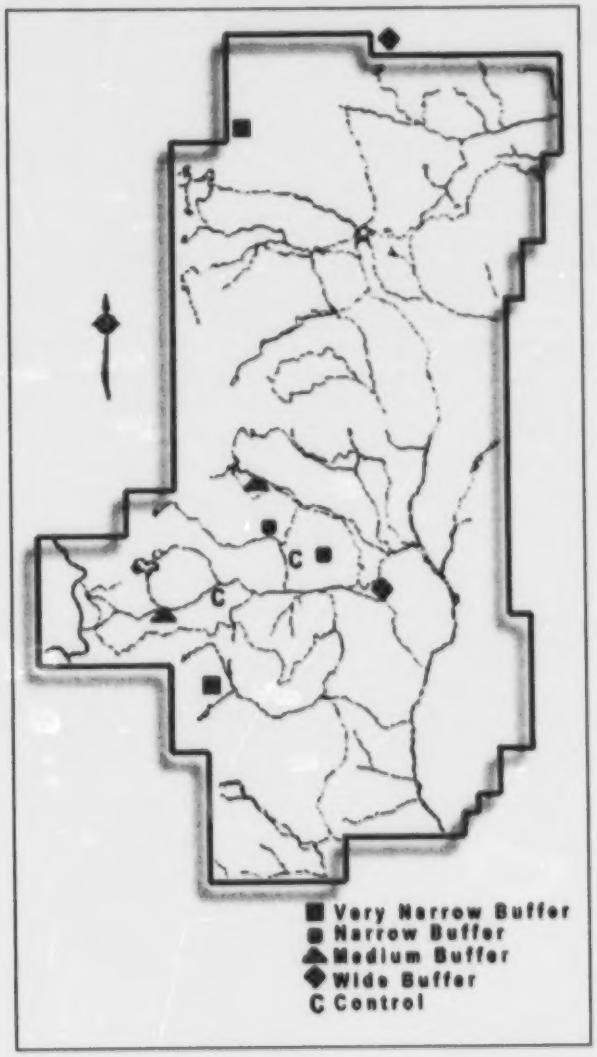

Figure 2. Location of study sites and treatment types with TFL 35.

Mean species richness and relative species abundance were calculated from the 1998 data. Richness was determined as the total number of species per treatment, averaged between replicates. Individuals were pooled between replicates to yield the total number of birds in each treatment. Relative abundance was determined for each treatment, expressed as the proportion of total individuals occupied by individuals of a given species.

Habitat and Vegetation Surveys

Habitat mapping and vegetation sampling were completed in 1998. Broad habitat features at each site were recorded onto the appropriate survey map. Mapped features included riparian buffer edges, root masses, wildlife tree patches, slash/blowdown piles, and steep slopes. Following the 1999 field season, habitat maps will be digitized and entered into a GIS database. Songbird territory locations will be overlaid to determine general species–habitat associations.

Vegetation sampling was conducted at a finer scale to quantify habitat structure and composition. Grids were divided into 4 strata; each stratum represented a given distance from the stream (0–25 m, >25–50 m, >50–80 m, >80–150 m). Three vegetation plots were systematically placed within each stratum for a total of 12 plots/grid. Circular plots of 10-m radius (0.03 ha), and line-intercept and point-intercept transects of 20-m length were used to sample vegetation attributes.

Circular plots were divided into 4 quadrants and number of trees was tallied by species and diameter class (Noon 1980). Diameter classes were: 10–20 cm (class 1), >20–40 cm (class 2), >40–60 cm (class 3), and >60 cm (class 4). All live trees >10 cm diameter at breast height (dbh) and all standing snags >10 cm dbh and >1.5 m height were included. A representative tree from each diameter class was selected and the height measured with a clinometer and hip chain. A convex densiometer was used to measure canopy cover along cardinal directions at the plot centre (Martin et al. 1997).

Line-intercept transects (Cook and Stubbendieck 1986) were used to measure shrub cover by species, height class, and vigour. Two line transects were placed in each plot, oriented at right angles to each other and intersecting at the plot centre. Shrub height was divided into 4 classes: 0–1 m (class 1), >1–3 m (class 2), >3–5 m (class 3), and >5–8 m (class 4). Vigour was described as excellent, good, poor, and dead. The point-intercept method (Bonham 1989) was used to estimate ground cover by height class and cover type. A

wooden rod, equipped with a sharpened metal end and a carpenter's level, was dropped at 2-m intervals along both line transects. Ground cover height was divided into 5 classes: 0–0.10 m (class 1), >0.10–0.25 m (class 2), >0.25–0.50 m (class 3), >0.50–1.0 m (class 4), and >1.0 m (class 5). Ground cover types were defined as: herb, grass, moss, lichen, coarse and fine woody debris, litter, bare ground, rock, livestock scat, and water.

Principal components analysis is being used to determine the structure of the vegetation data and to reduce the number of vegetation variables. Important variables will be tested for differences between treatments. To date, descriptive analyses have focused on canopy composition, tree diameter distribution, percent shrub cover by height class, and percent ground cover by cover type.

Habitat Associations

Habitat association data will be collected from May to August 1999 and sampling methods are in preparation. The goal is to further investigate the biology and habitat requirements of particular species. Detailed information on habitat use will be combined with vegetation plot and habitat mapping data to determine species' habitat preferences and structural requirements. Sampling will focus on several common species, including the winter wren, Wilson's warbler, and Townsend's warbler (see Table 1).

Data collection will involve behavioural observations and the construction of time budgets for selected species. The movements and activities of an individual will be recorded during a timed period, using a small tape recorder and microphone. Habitat type, specific vegetation structure used, and bird activity will be noted according to a standardized set of habitat, structure, and activity classes. Time budgets will address the proportion of an individual's time spent in each class.

RESULTS

Breeding Birds

Twenty-one songbird species were recorded during the 1998 field season (Table 1). The medium treatment (mean buffer width = 32 m) had the greatest total species richness (13), followed by the control (10), wide (10), narrow (9), and very narrow treatments (5). The medium buffer also possessed both the highest mean species richness and the greatest variability in richness between replicates (Fig. 3). Control plots had the second highest mean richness while the lowest mean richness occurred in the very narrow treatment.

The total number of recorded individuals was relatively low across all treatments, ranging from 11 individuals in the very narrow treatment, to 34 individuals in the control. Community evenness was low across sites and samples were dominated by 1 or 2 common species. Dark-eyed juncos were a dominant species in all harvested sites, but were absent from the controls. Conversely, golden-crowned kinglets were recorded in high abundance only within the wide and control sites. The winter wren reached maximum abundance in the medium treatment and Townsend's and yellow-rumped warblers occurred in low numbers in all treatments except the very narrow buffer.

Table 1. Songbird species list for 1998.

Common name	Scientific name
American robin	*Turdus migratorius*
Black-capped chickadee	*Poecile atricapillus*
Boreal chickadee	*Poecile hudsonicus*
Brown creeper	*Certhia americana*
Chipping sparrow	*Spizella passerina*
Dark-eyed junco	*Junco hyemalis*
Dusky flycatcher	*Empidonax oberholseri*
Golden-crowned kinglet	*Regulus satrapa*
Hammond's flycatcher	*Empidonax hammondii*
Lincoln's sparrow	*Melospiza lincolnii*
MacGillivray's warbler	*Oporornis tolmiei*
Mountain bluebird	*Sialia currucoides*
Pine siskin	*Carduelis pinus*
Red-breasted nuthatch	*Sitta canadensis*
Savannah sparrow	*Passerculus sandwichensis*
Swainson's thrush	*Catharus ustulatus*
Townsend's warbler	*Dendroica townsendi*
Varied thrush	*Ixoreus naevius*
Wilson's warbler	*Wilsonia pusila*
Winter wren	*Troglodytes troglodytes*
Yellow-rumped warbler	*Dendroica coronata*

Community composition appeared to change with increasing riparian buffer width. The very narrow and narrow treatments contained several species commonly found in open or mixed habitats. Mountain bluebirds, savannah sparrows, and Lincoln's sparrows all occurred in at least 1 of the 2 treatments. As buffer width increased, forest-dwelling species became more prevalent. Red-breasted nuthatches were present in the medium, wide, and control sites, while

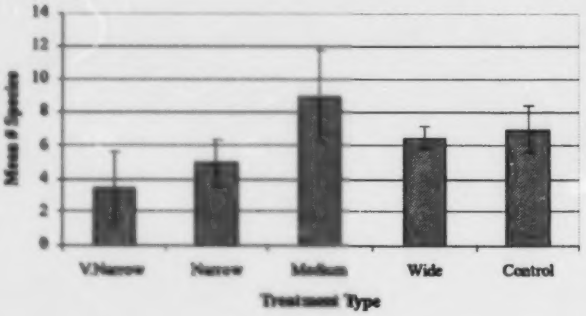

Figure 3. Mean avian species richness for July 1998 (±1 SE).

boreal chickadees occurred in the wide and control sites. Brown creepers, identified as mature forest specialists (Erlich et al. 1988), and varied thrush were recorded only within the control sites.

Vegetation

Similar tree species dominated the canopy across all treatments. Subalpine fir and hybrid white spruce each occupied >15% of the mean composition in each treatment. Snags comprised >25% of the canopy in all treatments except wide and control, where they were recorded at 10 and 15%, respectively. The very narrow, narrow, and medium treatments each contained 4 live tree species. The wide treatment included 6 tree species and the control included 5 tree species.

Tree diameter distribution was characterized for the first stratum in each grid (0–25 m from stream). Diameter class 1 trees (0–20 cm dbh) dominated the riparian canopy in the very narrow and medium treatments (Fig. 4). Diameter class 1 and 2 trees occupied approximately the same proportion of the riparian canopy in the narrow, wide, and control treatments. The highest mean number of large trees (>40 cm dbh) was recorded in the wide treatment and the control.

Shrub cover and number of understory layers appeared to change with riparian buffer width. The lowest overall shrub cover occurred in the very narrow treatment, while the highest overall cover was recorded in the wide and control sites. The greatest number of vertical understory layers was present in the control, with all 4 height classes recorded in each stratum. The lowest number of vertical layers occurred in the very narrow treatment, with only 2 height classes recorded. In the first stratum (0–25 m from stream), low shrubs (0–1 m) dominated riparian habitats at all sites; the highest mean cover occurred in the narrow treatment and control (Fig. 5). The remaining height classes increased in mean cover with increasing buffer width, reaching maximum cover in the wide treatment and declining in the control.

Ground cover and type also demonstrated changes across treatments. Mean herbaceous plant and moss cover declined with decreasing buffer width, while mean coarse woody debris and bare ground cover increased. Plant and moss cover were lowest across all strata in the very narrow treatment and highest across all strata of the control. In harvested sites, mean plant and moss cover decreased and bare ground increased with increasing distance from the stream. In the controls, mean plant cover decreased and moss cover increased from the riparian to the upland.

DISCUSSION

The distribution of songbird species occurring across treatments in this study is comparable to results cited by similar research elsewhere in the province. Gyug (1996) examined birds breeding in riparian and upland habitats under 3 harvest regimes (control, forest corridors, clearcuts) in the MS and ESSF zones of the Okanagan Highland Ecosection. Results indicated that several forest species were sensitive to harvesting, including the boreal chickadee, brown creeper, Townsend's warbler, and winter wren. Open-forest species included Wilson's and yellow-rumped warblers, while dark-eyed juncos were listed as generalists. A separate study in the ESSF recorded golden-crowned kinglets, Townsend's warblers, and winter wrens in controls and in stands adjacent to clearcuts; boreal chickadees and brown creepers were counted only in the control forests (Wetmore et al. 1985).

Though only a single survey was conducted in 1998, results support the literature in concluding that particular

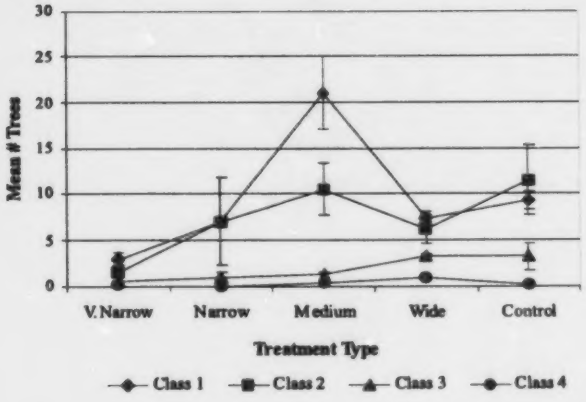

Figure 4. Mean number of trees (±1 SE), by tree diameter (dbh) class, across buffer widths for Stratum I (riparian strata).

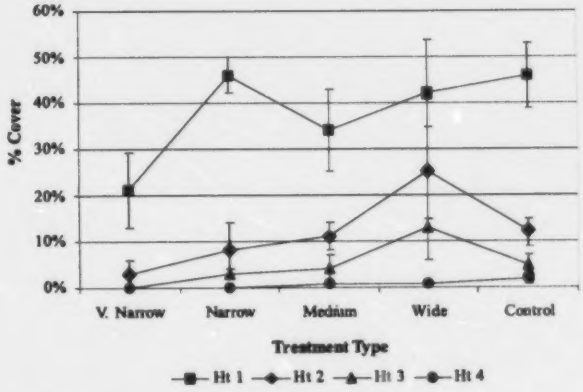

Figure 5. Mean percent cover (±1 SE), by height class, across buffer widths for Stratum I (0–25 m from stream).

species are forest specialists, while other species are either edge-tolerant forest birds or habitat generalists. Brown creepers and boreal chickadees appear the most sensitive to harvesting effects, indicating the need for undisturbed forests as breeding habitat. While golden-crowned kinglets, red-breasted nuthatches, Townsend's warblers, and winter wrens require forested habitat, they seem to tolerate forest openings, edges, and adjacent clearcuts. Dark-eyed juncos and American robins use both clearcuts and forest buffers. These species, and other generalists, are likely to dominate songbird communities in heavily harvested forests.

The greatest species richness was recorded in riparian buffers averaging 30 m wide, suggesting the possible existence of a threshold width. The presence of forest and clearcut in a relatively small area contributes to maximum richness by attracting forest, open habitat, and generalist species. Decreasing forest area results in a fewer forest species, while increasing forest area leads to fewer generalist and open habitat species. Whether riparian buffers of medium width can support disturbance-sensitive forest species has yet to be determined.

Many riparian studies have indicated that wider reserve zones are required to maintain songbird richness and diversity at or near predisturbance levels (Triquet et al. 1990, Keller et al. 1993, Darveau et al. 1995, Dickson et al. 1995). Spackman and Hughes (1995) assessed stream buffer widths in Vermont and found that minimum widths of 75–175 m were necessary to include 90% of all riparian-associated birds. Croonquist and Brooks (1993) investigated riparian bird communities in central Pennsylvania and discovered that an undisturbed reserve zone >25 m in width was required to support sensitive species. A study in eastern British Columbia predicted that prescribed reserve zone widths would likely possess lower densities of total and riparian-associated birds than the 70-m buffers recommended by the research (Kinley and Newhouse 1997).

The current riparian management guidelines in British Columbia may be inadequate to maintain the value of stream riparian systems to forest songbird species. Conversely, assigned reserve widths may be sufficient to support forest species and wider reserves may be unnecessary. Studies have shown that, while all forest-interior species were not retained, bird abundance did not decrease significantly with decreasing buffer width (Thurmond et al. 1995) and buffer strips could still provide breeding habitat for many songbird species (Johnson and Brown 1990).

The high elevation forests in the study area are historically disturbance-dependent, relying on stand-initiating fire and insect outbreaks (B.C. Ministry of Forests, and BC Environment 1995*b*). Songbird species breeding in these forests may also display some disturbance adaptation and ability to adjust to harvesting and other human impacts. Species commonly considered as forest specialists, including the Townsend's warbler and varied thrush, may possess greater flexibility in their selection of breeding habitats. Conversely, due to the short breeding seasons and limited resources often experienced in these high elevation sites, forest birds may experience increased sensitivity to harvesting and habitat loss. Species able to tolerate edge environments at lower elevations may require relatively large tracts of undisturbed forest at higher elevations. Given this uncertainty, the response of forest birds to harvesting in high elevations requires further study, both at the community and species levels.

MANAGEMENT IMPLICATIONS

As research is ongoing, few management recommendations can be discussed now. When interpreting results and their implications to management, community composition must also be considered in addition to species richness. While the highest richness may occur in a buffer of 30 m, the addition of several open habitat species may occur at the expense of few forest specialists. Thus, management should not necessarily aim to maximize species richness, but to provide for those species most vulnerable to harvesting disturbance.

ACKNOWLEDGEMENTS

I thank the following agencies and companies for their funding and/or administrative support: Natural Science and Engineering Research Council of Canada (NSERC), Forest Renewal British Columbia (FRBC), Science Council of British Columbia (SCBC), Weyerhaeuser Canada Ltd., and British Columbia Conservation Foundation (BCCF). In addition, I thank thesis supervisors Dr. C. Prescott and Dr. T. Dickinson, and committee members Dr. J. Richardson and Dr. J. N. M. Smith for their advice and patience.

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Grazing Impacts on the Biodiversity of Riparian Ecosystems

Astrid M. van Woudenberg
Cascadia Natural Resource Consultants Inc.
2675 Skeena Drive, Kamloops, BC, V2E 2M9, Canada
astrid@cnrc.com

ABSTRACT

The first phase of a multiyear research project to investigate the effects of grazing on the biodiversity of grassland riparian ecosystems was completed in 1992–97 with the pretreatment (grazing) sampling period. The study area is in the Thompson Dry Cool Interior Douglas-fir Variant Grassland Phase (IDFdk1a) at Hamilton Commonage near Merritt, B.C. Pretreatment (grazed) inventories were sampled for bird, small mammal, herpetofauna, and vegetation communities at 6 ponds in Hamilton Commonage. The research design included 5 exclosures at 4 ponds, with 2 unfenced ponds (control). Pretreatment inventory results suggested that historical and some current grazing impacts have reduced the woody component in riparian habitats. Regeneration of woody plant species was not observed. Pretreatment inventory results of wildlife communities indicated that the structure of shrub-aspen complexes is important for species richness. Several species, particularly birds, were associated only with shrub-aspen habitat types. Open habitat types that did not have the structural complexity of woody riparian vegetation tended to have a different species composition, including more generalists. Overall, riparian habitat was found to be utilized disproportionately by wildlife. Prescriptions for grazing intensity and seasonal schedules specific to riparian vegetation may be a sustainable management alternative.

Key words: grassland, livestock grazing, riparian.

Riparian habitats are critical to many species of wildlife (Skovlin 1984, Schulz and Leininger 1990), particularly in dry, grassland ecosystems (Thomas et al. 1979, Kauffman and Kreuger 1984). Riparian ecosystems provide shade, a variety of quality forage, and drinking water in arid or semiarid environments (Cross 1985, Bunnell and Rautio 1995). The gentle topography offers easy access, and the combination of habitat characteristics strongly attracts cattle to riparian areas, where they can remain after sufficient forage has been depleted (Roath and Krueger 1982, Skovlin 1984, Gillen et al. 1985). Consequently, grazing impacts by domestic livestock can reduce graminoid, forb, and shrub cover (Szaro and Pase 1983, Clary and Medin 1990), which can result in loss of habitat suitability for specific wildlife species (Medin and Clary 1989, 1990, Bunnell and Rautio 1995).

Grazing management regimes are usually based on upland plant community requirements and are often inappropriate for riparian species (Skovlin 1984, Elmore 1992). For example, the multipasture rest-rotation system commonly used is sustainable for upland plant species, but is inappropriate for many riparian habitat types (Elmore 1992). During the late, dry summer months, utilization of riparian herbaceous vegetation can reach 80–90% by the time the target of 60% use of upland plant species has been achieved. Cattle use riparian herbs and forbs when upland grasses have dried and seeded (Skovlin 1984, Clary and Booth 1993). At 45% use of riparian herbaceous component, cattle begin to utilize the first year's growth of the riparian shrub component, primarily willow species. At 90% riparian herbaceous utilization, cattle have consumed previous years' growth of willow. The grazing strategy is designed for the sustainability of nonwoody plants and cannot be applied on a long-term basis to riparian communities with woody plant components.

Reduced shrub structure in riparian communities can alter or reduce critical habitat features for wildlife, resulting in shifts in community composition (Conine et al. 1978, Kauffman and Kreuger 1984). Few studies have documented the effects of livestock grazing on biological diversity in riparian habitats in British Columbia (Bunnell and Rautio 1995). Specifically, grazing impacts on grassland riparian habitats and associated wildlife species in the Thompson-Nicola subregion of British Columbia have not been documented until now. The region has a strong livestock industry, which has been growing since the late nineteenth century (D. Low, B.C. Ministry of Environment, Lands and Parks, pers. comm.; B. Wikeem, B.C. Ministry of Forests, pers. comm.). No grassland areas that have not received some grazing impact are available for a comparative study in the Thompson-Nicola region (I. Barnett, Ducks Unlimited, pers. comm.; B. Wikeem, B.C. Ministry of Forests, pers. comm.;

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

L. Leach, B.C. Cattlemen's Association, pers. comm.; J. Lauder, Lauder Ranch, pers. comm.; L. Guichon, Gerrard Guichon Ranch, pers. comm.).

As part of a long-term research program, the impacts of grazing on the biological diversity of grassland riparian ecosystems will be investigated to delineate integrated management strategies sustainable for indigenous riparian ecosystems. The pretreatment inventory stage of the program was completed in 1997. Birds, small mammals, herpetofauna, and vegetation were sampled in riparian and upland habitats of study ponds at Hamilton Commonage, near Merritt, B.C. The experimental treatment (exclosures surrounding riparian habitats) was applied and will remain under posttreatment investigation for the next several years. As the rate of change in species composition slows or stabilizes, indigenous community composition may be realized; experimental grazing strategies may then be tested for their ability to sustain species composition achieved with fencing.

STUDY SITE

Hamilton Commonage is Crown land located in the Merritt Forest District and Thompson-Nicola subregion (between 50°00' and 50°30' N, 120°00' and 121°00' W; Fig. 1). The Commonage became a government reserve in 1887 for protection from alienation (G. Hayes, B.C. Ministry of Forests, pers. comm.; D. Jury, B.C. Ministry of Environment, Lands and Parks, pers. comm.). The study site elevation range is 1,150–1,250 m, and it is in the IDFdk1a biogeoclimatic site unit, known as the Thompson Dry Cool Interior Douglas-fir Variant Grassland Phase (Lloyd et al. 1990). Mean annual precipitation for the IDFdk site unit is 438 mm (range 376–512 mm; Lloyd et al. 1990).

Typically referred to as the "upper grasslands," IDFdk1a grassland ecosystems are distinguished from the Bunchgrass zone by the lack of a distinct upland shrub layer (Lloyd et al. 1990). The upland communities of the commonage are dominated by stiff needlegrass, with a Kentucky bluegrass subcomponent, and have a sparse shrub layer of pasture sage and rabbitbrush. In riparian communities, redtop and foxtail barley are typically the dominant graminoids, and wire rush occurs most abundantly closest to the water's edge. Beaked sedge and bulrush comprise emergent vegetation. Aspen riparian communities typically have Nootka rose and common snowberry present in the shrub layer. Nuttall's alkaligrass and alkali saltgrass characterize alkaline soils. Scientific names for all species observed during the study are listed in Appendix 1.

Hamilton Commonage has been grazed since the 1870s (G. Hayes, B.C. Ministry of Forests, pers. comm.). In addition to cattle grazing, winter horse grazing during the 1930s and 1940s was particularly damaging to the upper grasslands. Relatively uncontrolled grazing continued until the 1950s and 1960s, when management became somewhat stricter under the Grazing Act of 1917. Today, Hamilton Commonage is grazed by local permit holders. The study ponds fall within the pastures used for livestock grazing by Douglas Lake Ranch and Gerard Guichon Ranch. The overall utilization of Hamilton Commonage is approximately 55% annually.

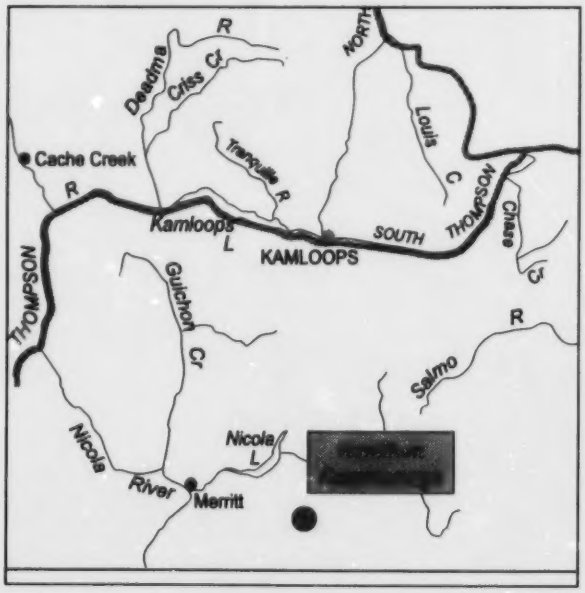


Figure 1. Study site at Hamilton Commonage.

METHODS

Bird Communities

Grids (20 X 20 m) were placed around ponds parallel to the shoreline, including riparian and upland habitats in a 10 to 1 ratio, to sample bird species abundance and use of habitat features, similar to spot-mapping (Hamel 1984). Most riparian borders were ≤10 m wide. Waterfowl, shorebirds, raptors, and passerine birds within the riparian border and within 50 m of upland were recorded and mapped (sampling rate: 100 m/6 min; Manuwal and Carey 1991). For details of this and other study methods, see van Woudenberg (1997).

Small Mammal Communities

Presence of small mammals was sampled using mark–recapture techniques in a 6 X 8 m rectangular grid with Longworth live-traps (T. Sullivan, University of British Columbia, pers. comm.). The rectangular grid oriented lengthwise along the shoreline provided greater sampling effort along the longer dimension of the riparian border and ecotone. Sampling was done at least twice at each study pond during summer and fall. Standard data recorded from captured animals included sex, weight, and reproductive

status, as well as vegetation species and structure.

Herpetofauna Sampling

A combination of pitfall traps and time-constrained searches was used in 1993 to inventory herpetofauna (Block et al. 1988). Grids (6 X 8 m) of 36 6.9-litre traps were placed at each pond within the riparian zone and adjacent upland (Corn and Bury 1990). However, due to extensive damage to traps by cattle trampling, inventory was restricted to nocturnal auditory surveys.

Vegetation Sampling

A minimum of 3 permanent transects was established in distinct riparian vegetation types at each pond, including shrub-aspen complexes, grass-shrub, and grass-forb. Locations were recorded using GPS (global positioning system) and marked in the field with rebar at either end of each transect for long-term monitoring. Line-intercept method was used for transects that traversed woody plant species habitats.

Daubenmire frames (20 X 50 cm plots) were placed within each distinct riparian band (Daubenmire 1959). Plots were oriented lengthwise, perpendicular to the water's edge, with 1 plot per narrow riparian band, and 3–5 equidistant plots in wider bands (>5 m). In the upland vegetation zone, 3 plots were placed along each transect. Species composition and structure were sampled by recording percent cover within each height strata (0–10 cm, 10–30 cm, 30–60 cm, 60–100 cm, and >100 cm); both live and dead vegetation were recorded. The data provided a volume estimate of vegetation biomass. Percent of bare ground was also recorded.

Exclosures

Exclosures were delineated by pairing similar areas for grazed and ungrazed treatments, based on similar historical utilization, plant species, and topography. Five exclosures were constructed at 4 study ponds in 1995. The first year of posttreatment effect was 1996.

Data Analysis

The Shapiro-Wilk test ($\alpha = 0.2$) and the normal distribution *G*-statistic test ($\alpha = 0.05$; Zar 1995) were used to test the vegetation and bird data for normal distributions. Most data were found to be not normally distributed, resulting in the application of nonparametric statistics.

Shannon-Weaver and Simpson diversity indices were calculated for bird communities, as well as riparian and upland vegetation communities sampled at each pond. *T*-tests were used to compare species richness indices between riparian and upland (and transition, where sampled) vegetation species (Sokal and Rohlf 1981). An analysis of variance was performed on the Shannon-Weaver diversity indices calculated for bird communities to compare species richness between ponds and years.

Diversity of riparian and upland vegetation communities was compared using the Wilcoxon nonparametric test for differences in the species richness. The Wilcoxen test can increase the chance of a Type II error, so $\alpha = 0.1$ was selected. Transition vegetation was not included in the analysis because sample sizes for each pond were small and the delineation of the transition zone tended to be subjective.

Ordination was used to determine upland and riparian community differences in the bird and vegetation data (Magurran 1988). By collapsing the data into a 2-dimensionality (x-y axes), species richness could be compared between riparian and upland communities without the masking effects of species distributions (Krebs 1989). Ordination analysis was used to classify species by their occurrence in only 1 zone (riparian, transition, or upland); species occurring at least once in all zones were assigned to a common category. Ordination analysis was also performed on the bird activity data to determine if specific activities of species were associated with particular habitat zones.

Riparian and upland vegetation communities were tested for significant compositional differences using Principal Components Analysis (PCA). A correlation matrix was used to calculate the eigenvalues to detect the natural associations among the variables (species). The dimensionality of the data was reduced using PCA to differentiate communities using few, new variables that were based on combinations of the many variables sampled.

RESULTS

For detailed results, the Project Phase I report should be consulted (van Woudenberg 1997).

Vegetation Community Analysis

Data at all ponds were not normally distributed ($\alpha = 0.05$). An α-level of 0.2 was selected for the Shapiro-Wilk test for normality to reduce the chance of a Type II error, necessitated by the variability in sample sizes between ponds and the inherent environmental variation among ponds (Zar 1995).

Cumulative Frequency Distributions

Cumulative frequency distribution curves were plotted to indicate the distribution and composition of species at all ponds in riparian, upland, and some transition zones documented in each year of sampling. The common dominant riparian species were Kentucky bluegrass, common dandelion, Nuttall's alkaligrass, Baltic rush, foxtail barley, redtop, and beaked sedge; common dominant upland species were June grass, pussytoes, Kentucky bluegrass, yarrow, and common dandelion (scientific names are listed in Appendix A). Transition zones were dominated by riparian and upland species.

Table 1. Principal Components Analysis correlation matrix results of principal indicator species for riparian and upland zones for Hamilton Commonage, 1995. Scientific names are listed in Appendix A.

Pond	Riparian zone species	Upland zone species
HC01	Purple-leaved willowherb	Slender wheatgrass
	Unknown moss spp.[a]	Unknown pussytoes spp.
	Canada thistle	Showy daisy[a]
	Redtop	Field locoweed[a]
	Sloughgrass	Timber milk-vetch[a]
	Foxtail barley	Unknown Carex spp.[a]
HC02	Foxtail barley	Yarrow
	Unknown fungus spp.	Unknown pussytoes spp.
	Field mint	Kentucky bluegrass[a]
	Early blue violet	Unknown vetch spp.[a]
	Canada fleabane	Stiff needlegrass
HC07	Purple-leaved willowherb	Shaggy fleabane
	Small bedstraw	Junegrass
	Beaked sedge	Cut-leaved daisy
	Field mint	Vetch spp.
	American brooklime	Needle-and-thread grass
	Silverweed[a]	
HC10	Baltic rush[a]	Kentucky bluegrass
	Field mint	Hillside milk-vetch
	Beaked sedge	Field pussytoes
	Foxtail barley	
	Silverweed	
	Bitter fleabane[a]	
HC15	Common dandelion	Stiff needlegrass
	Purple reed grass	Rosy pussytoes
	Silverweed	
	Field mint	
	Foxtail barley	
	Shore buttercup	
HC29	American brooklime	Common dandelion[a]
	Purple-leaved willowherb	Yarrow[a]
	Short-awn foxtail	Junegrass
	Water smartweed	Unknown pussytoes spp.[a]
	White water buttercup	

[a] Species occurred in all zones at that pond according to ordination analysis, but was more likely to occur in the indicated zone.

Species Diversity

Shannon-Weaver diversity indices for vegetation showed that riparian communities had the highest species diversity, on average, compared with upland communities at study ponds. Simpson's diversity index results tended to be similar to Shannon-Weaver index results for all years and study ponds.

T-test results of the Shannon-Weaver diversity indices showed significant differences ($\alpha = 0.05$) between riparian and transition, riparian and upland, and transition and upland communities at all ponds except between riparian and transition at HC29 (1995) and HC15 (1993).

Ordination Analysis

The ordination analysis of species richness for the 1995 data (Fig. 2) showed that riparian communities at all ponds, except HC10 in 1995, had higher species richness than upland communities. Overall, species common to both riparian and upland areas showed the second highest level of richness. Species unique to each zone, and species common to more than 1 zone were identified. The data were not distributional since each species is only categorized according to its location, not its frequency in any 1 particular zone. The transition zone was included only in the analysis for all ponds combined, since individual sample sizes for individual ponds were too small.

Principal Component Analysis

PCA results for the 1995 data indicated a clear distinction between riparian and upland vegetation communities. The 1994 data set did not segregate among the principal axes as cleanly, but showed significantly different trends between the riparian and upland species composition. The segregated distribution or "loadings" of the data points at opposite ends of an axis indicated riparian and upland species were well-segregated. Table 1 shows the indicator species that accounted for the significant segregation of points. The presence of 1 of these species in a vegetation plot was a reasonably accurate indicator of habitat type.

PCA results for slope analysis showed that slope was not significantly related to vegetation species composition.

Vegetation Structure

Figures 3a and 3b show the structural composition of vegetation in riparian and upland zones. Dead vegetation was less abundant than live vegetation in most height strata at most ponds; the trend was least pronounced in the first height stratum, particularly in the upland, in both years at most ponds. Dead vegetation exceeded the abundance of live vegetation in the first height stratum at HC10, HC15, and HC29 in the upland in 1994, and at HC01, HC02, HC10, and HC15 in the upland in 1995; at HC10 the dead vegetation composition also exceeded live in the lowest height stratum of the riparian zone.

Less difference occurred between live and dead upland vegetation than between live and dead riparian vegetation. Live vegetation had more composition in all layers overall than dead vegetation did.

The riparian community was more structurally complex than the upland, with greater composition in the higher strata levels (60–100 cm and >100 cm). Vegetation in the upland zone never exceeded 60–100 cm.

Figures 4a and 4b indicate the occurrence of physical features, including livestock scat, bare ground, rock, and coarse

woody debris (CWD), that were present in riparian and upland zone plots in 1995. Half of the ponds had higher amounts of bare ground in the upland, and half had more bare ground in the riparian zone. Livestock scat was higher in the upland than it was in the riparian. The amount of rock was higher in the riparian than in the upland at HC07 and HC29; all other ponds had more rock present in upland than riparian plots. CWD was more abundant, overall, in the riparian than in the upland zone.

The line-intercept data were used to determine the proportion of total detections of each species in each specific growth or decomposition stage (i.e., CWD, snag/dead shrub, old growth, mature, juvenile, sapling, seedling). At HC02, CWD was the most common stage detected and it had accumulated only from aspen. Gooseberry was the most commonly detected shrub, encountered most frequently as old growth, and secondly as mature or midseral development stage. Aspen seedlings and mature prickly rose were detected at moderate frequencies; Bebb's willow was encountered the least frequently and was present only in a mature stage.

At HC01 and HC29, aspen as CWD was also most frequently encountered. Aspen was detected at HC01 and HC29 more frequently in more stages of development than it was at HC02; this included snags, which were not present at HC02. Other species detected were shrubs, present in mature to older developmental stages. The seedling stage was not detected for any species other than aspen at all 4 ponds. HC07 was the only pond where the juvenile development stage was encountered for gooseberry, raspberry, and aspen. The most commonly encountered species at HC07 was Bebb's willow in the form of CWD; aspen in the same form followed closely. Old-growth Bebb's willow was the next most commonly detected, followed by mature and sapling stages of raspberry, prickly rose, Nootka rose, Mackenzie's willow, and gooseberry.

Table 2 indicates the amount and size of CWD detected at the 4 ponds in Hamilton Commonage in 1995. Generally, larger pieces of CWD, and no snags, were detected at HC01 and HC02, likely due to fallen snags, which were often suspended above the ground on top of other vegetation or CWD.

Figures 5a–d show what proportion of the total transect length was occupied by each shrub and trees species. At HC02, where the lowest number of species was documented, CWD and old-growth aspen occupied most of the area sampled along the transect. Gooseberry was also fairly common. Bebb's willow was present in the least amounts. Aspen also occupied the most area at all other ponds, although Bebb's willow was subdominant at HC07, with very little abundance

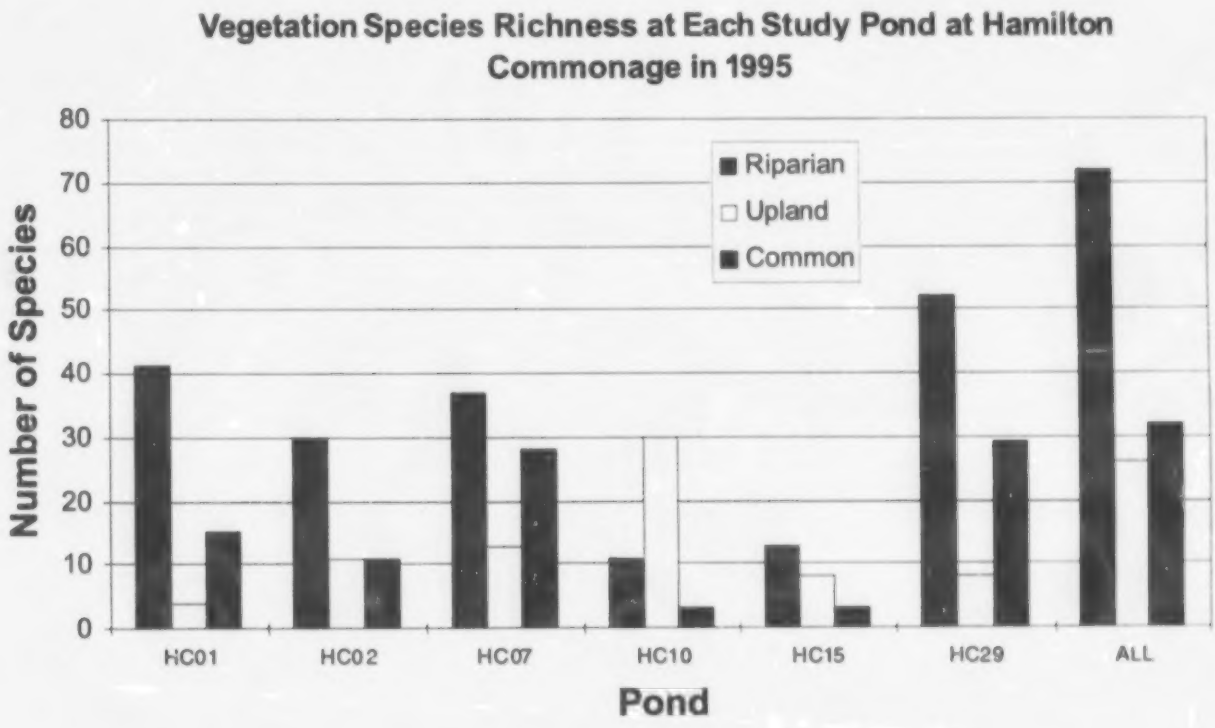


Figure 2. Ordination analysis comparing riparian and upland vegetation species richness.

at HC01 and HC29. Compositional distribution of the remaining, less common species, was similar to the trends in which they were detected.

Mature aspen was the tallest species documented at all ponds. Old-growth Bebb's willow was the second-tallest species documented at HC29 and HC07. At HC01, snowberry, gooseberry, and prickly rose were encountered at similar mean heights in the old-growth stage. Similarly, at all other ponds, mean heights of less abundant species were <2 m.

Bird Community Analysis

Cumulative Frequency Distributions

Cumulative frequency distribution curves were plotted for bird communities at all ponds in Hamilton Commonage for the 3 years of inventory. The distribution curves indicated that at all ponds for all study years, communities tended to be dominated by a few species. This was particularly the case for HC10 and HC15, and was least pronounced for HC07. The most commonly detected species included

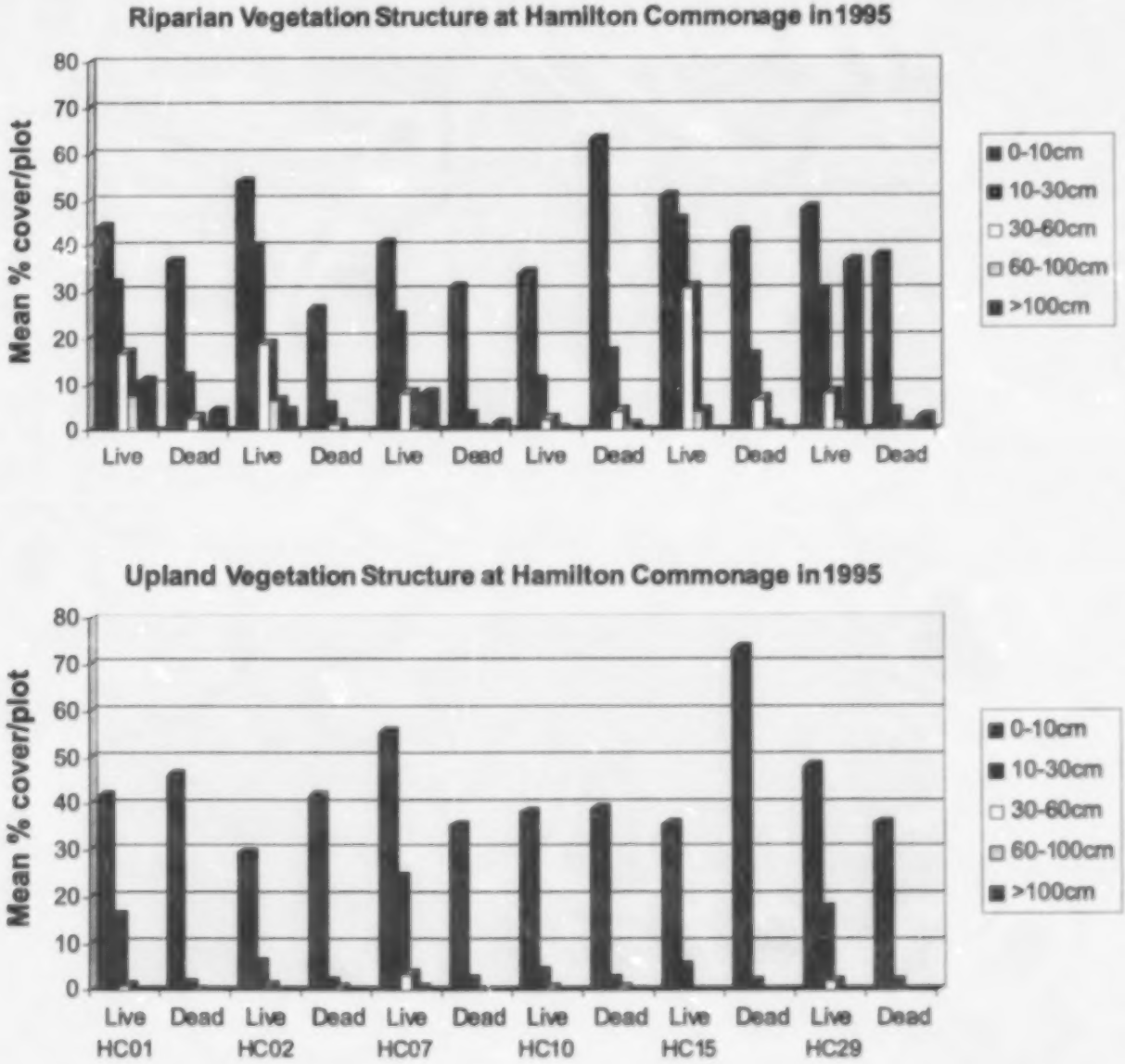


Figure 3a–b. (a) Riparian and (b) upland vegetation at Hamilton Commonage.

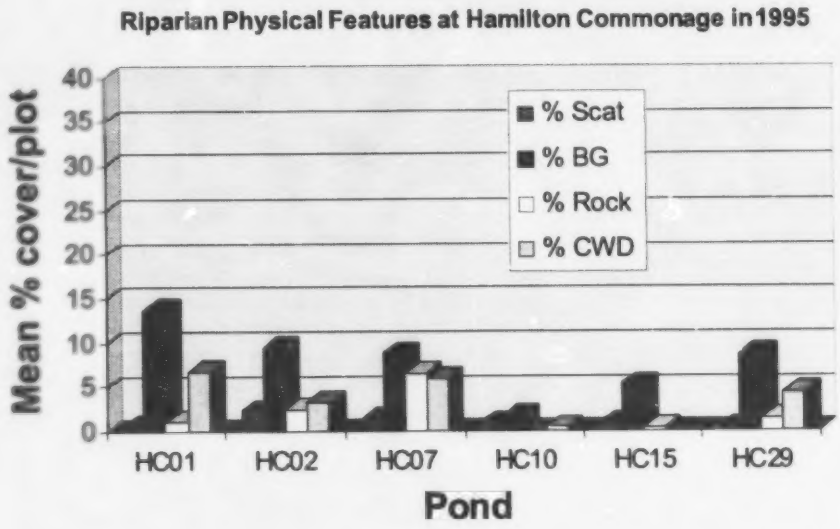
savannah sparrow, song sparrow, tree swallow, yellow-headed blackbird, eared grebe, European starling, ruddy duck, and lesser scaup (scientific names in Appendix 1).

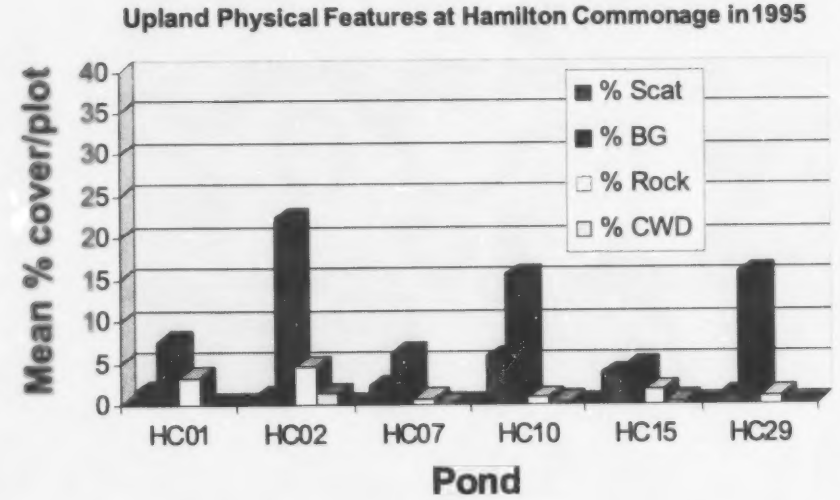
Diversity Indices

The highest number of species was consistently recorded at HC07 for each year's data. Of the large ponds, HC10 showed the least species diversity, although a high incidence of individuals was reported. An analysis of variance of the Shannon-Weaver diversity indices resulted in no significant difference in bird community diversity at each pond between years of sampling, but ponds differed significantly ($\alpha = 0.05$) in species diversity.

Ordination Analysis

Ordination results for the 1995 bird communities at Hamilton Commonage showed that at HC01 the greatest species richness occurred in the riparian and riparian/water





Figures 4a–b. Bare ground, livestock scat, coarse woody debris, and rock recorded in (a) riparian and (b) upland habitats at Hamilton Commonage in 1995.

Table 2. Coarse woody debris size at Hamilton Commonage, 1995.

Pond	Mean height (m)	Standard deviation	Range(m)	No. of pieces
HC29	0.19	0.254	0.02–1.3	58
HC07	0.12	0.103	0.02–0.6	117
HC02	0.28	0.155	0.08–0.7	23
HC01	0.41	0.805	0.02–6.3	77

zones. Species found only in the riparian zone included shorebirds, blackbirds, swallows, and Wilson's flycatcher; species found exclusively in the upland zone include American crow and vesper sparrow. Several waterfowl species and brown-headed cowbirds occurred in all zones. Species detected exclusively in the riparian zone at HC02 include the marsh wren, American kestrel, dusky flycatcher, mountain bluebird, northern oriole, and an unidentified sparrow. Similar species were also detected only in the riparian zone at HC07, including warblers and several other species. HC07 and HC29 had the highest species richness in the riparian zone. At HC10 the number of species detected exclusively in the riparian zone was equal to the number found in the upland zone. HC15 was the only pond where birds were not detected exclusively in the riparian area; this was the smallest pond. Overall, the highest species richness in Hamilton Commonage was detected in the riparian zone, excluding the overlap with the water zone. The category where birds were detected in the riparian, upland, and water zones together had the next highest species richness. The upland zone, overall, had the least species richness.

Bird Habitat

Ordination analysis was also used to show what activity the birds were engaged in when detected, and in which associated vegetation species and structure they were observed. Ehrlich et al. (1988), Peterson (1990), and Godfrey (1986) were used to determine the expected habitat feature associated with a critical life requisite or related activity. The results showed that most passerine species utilized aspen or aspen-willow complexes; shorebirds tended to use open or grassy habitats; and waterfowl used a variety of habitats, but were primarily detected on the water surface.

The results of the ordination analysis for species' activities associated with each zone at all study ponds showed that foraging was rarely detected in the upland, and at some ponds it was not detected at all. The activities most often displayed by species observed in the upland included territoriality and security behaviour. Horned larks and vesper sparrows were exclusively observed in the upland, most commonly displaying territoriality. In the riparian zone, types of behaviour associated with foraging, security, territoriality, and breeding activities were commonly detected. Results were similar in the emergent zone. Species commonly observed in the riparian were killdeer, Brewer's blackbird and common snipe. Territoriality tended to be the common activity observed in all zones at most ponds, and security was also common to all zones.

Table 3. Common names and codes of species detected in Hamilton Commonage. Scientific names are listed in Appendix A.

Common name	Species code
Montane vole	MIMO
Meadow vole	MIPE
Deer mouse	PEMA
unknown shrew species (*Sorex* spp.)	SOREX
Meadow jumping mouse	ZAHU

Small Mammal Community Analysis

Figure 6 shows the small mammal community composition at Hamilton Commonage in 1995 and Table 3 provides common names and species identification codes. Meadow voles dominated riparian communities, and deer mice were dominant in the upland, with the exception of HC02, where the riparian community was dominated by deer mice and subdominated by meadow voles, and HC29, where meadow voles were the dominant riparian species and montane voles dominated the upland. Transition zones that were wide enough to accommodate a line of traps were dominated by deer mice at HC01 and montane voles at HC07. Jumping mice were recorded in the riparian habitat at HC29, and shrews were caught at HC07. Montane voles were detected only at HC01 in the riparian zone and were present in the upland at all sampled ponds (except HC29) as a rare species; montane voles were also subdominant in the transition zone at HC01.

Small mammal trapping occurred at dawn and dusk. All species were caught most frequently in the morning trapping session. Of the few species caught in the evening, meadow voles were the most common.

Petersen's index values were calculated for closed mark–recapture procedure to estimate populations as each pond (Krebs 1989); HC01 was 140, HC02 was 29, and HC07 was 45 animals in 1995. The highest population estimate was made for HC01. Only sampling sessions during July and mid-August provided large enough sample sizes to calculate Petersen's population index. Each index is based on 4 grids within 1 sampling session. HC29 did not have a large enough sample size for calculation.

Sexes were not equally represented at ponds. Overall, females tended to be more abundant and weighed more than males. Females tended to be caught more frequently than males in riparian zones, and both sexes caught in riparian habitats tended to have higher weights than those caught in upland habitats. Transition zones showed intermediate

weights between riparian- and upland-caught animals. At HC07, only males were caught in the transition zone and more females were caught in riparian habitat. At HC02, no males were caught in the transition zone; females were abundant in the transition and riparian zones. Only females were caught in the emergent zones at HC02 and HC07; their weights were higher than riparian females at HC02, and lower than riparian females at HC07.

In the riparian zone, redtop was the most common species that dominated the vegetation community immediately surrounding the traps. Wire rush was the most frequently documented dominant species at traps where deer mice were

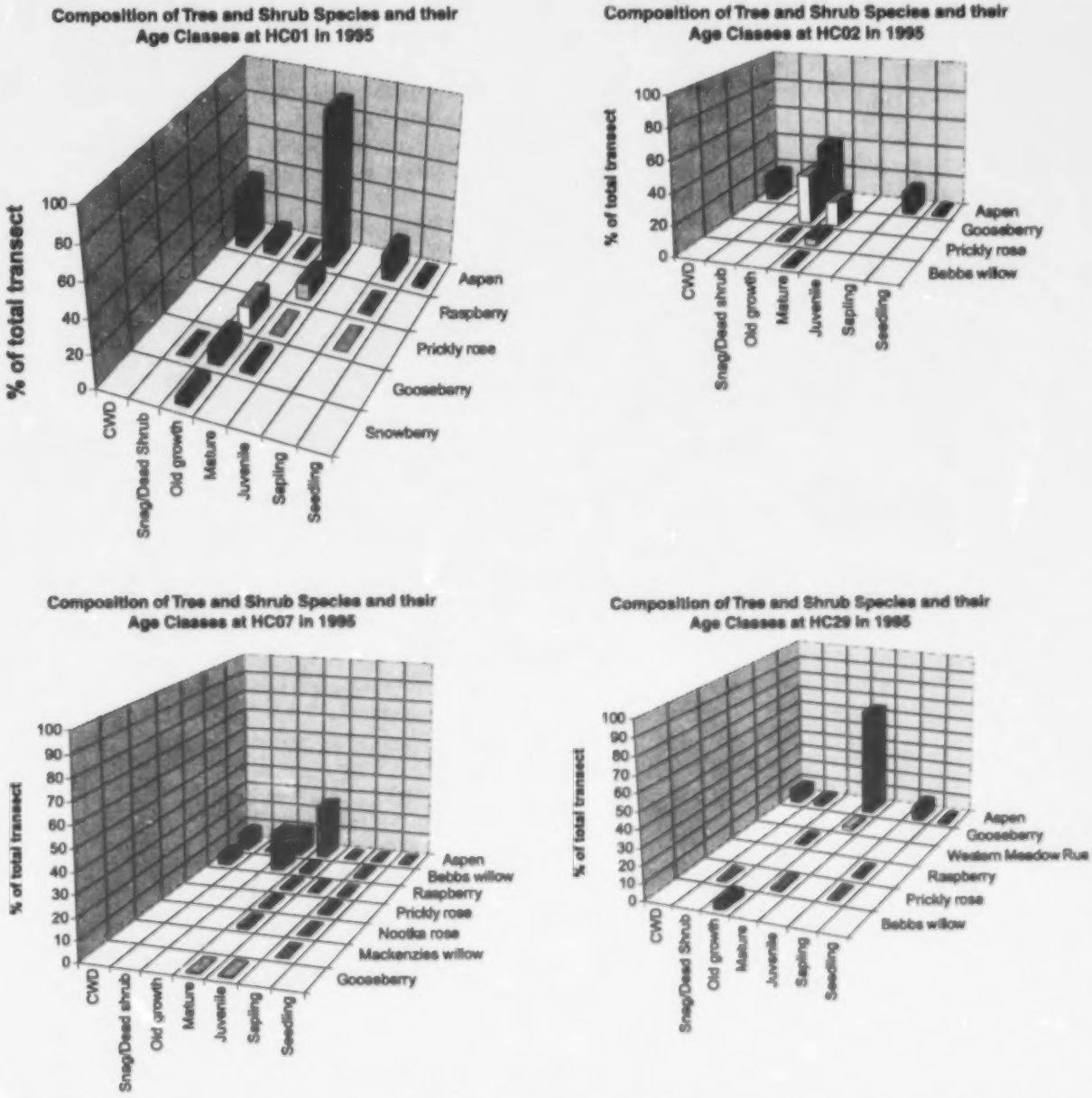


Figure 5a–d. Composition of shrub and tree species and age classes observed in line-intercepts at riparian areas sampled at (a) HC01, (b) HC02, (c) HC07, (d) HC29.

captured. Subdominant species included Nuttall's alkali grass, redtop, Kentucky bluegrass, wire rush, creeping spike rush, foxtail barley, and dandelion. Kentucky bluegrass was the dominant vegetation species recorded for all 3 small mammal species detected in the upland. Dandelion and stiff needlegrass were frequently subdominant species. Pussytoes, yarrow, cut-leaf fleabane, and field locoweed were frequently detected as 1 of the 3 subdominant species at traps where deer mice were caught. Subdominant species frequently detected at meadow and montane vole traps were showy fleabane, and yarrow and *Agoseris* species, respectively.

Overall, vegetation height was considerably greater at traps with animals in riparian zones than at those in upland zones. In riparian zones, vegetation at traps where meadow voles were caught was taller than at traps where other small mammal species were caught. In upland zones, vole species had similar vegetation structure that was often twice as tall as that recorded at traps where deer mice were caught. Variability of vegetation height data was fairly similar between all species within each zone, and was lower in the upland than the riparian zone. Vegetation height did not show any trends distinguishing breeding from nonbreeding animals.

Mean percent cover of bare ground was higher for deer mice habitats than for vole habitats in both riparian and upland zones. The amount of bare ground recorded tended to be higher for nonbreeders than breeders in both riparian and upland zones. Overall, the least abundance of bare ground was recorded at traps in which meadow voles were caught in riparian zones.

CWD was absent from all upland habitats and did not occur at traps in riparian zones in which montane voles were caught. The highest abundance of CWD, by far, was recorded at traps that caught meadow voles in riparian zones. A

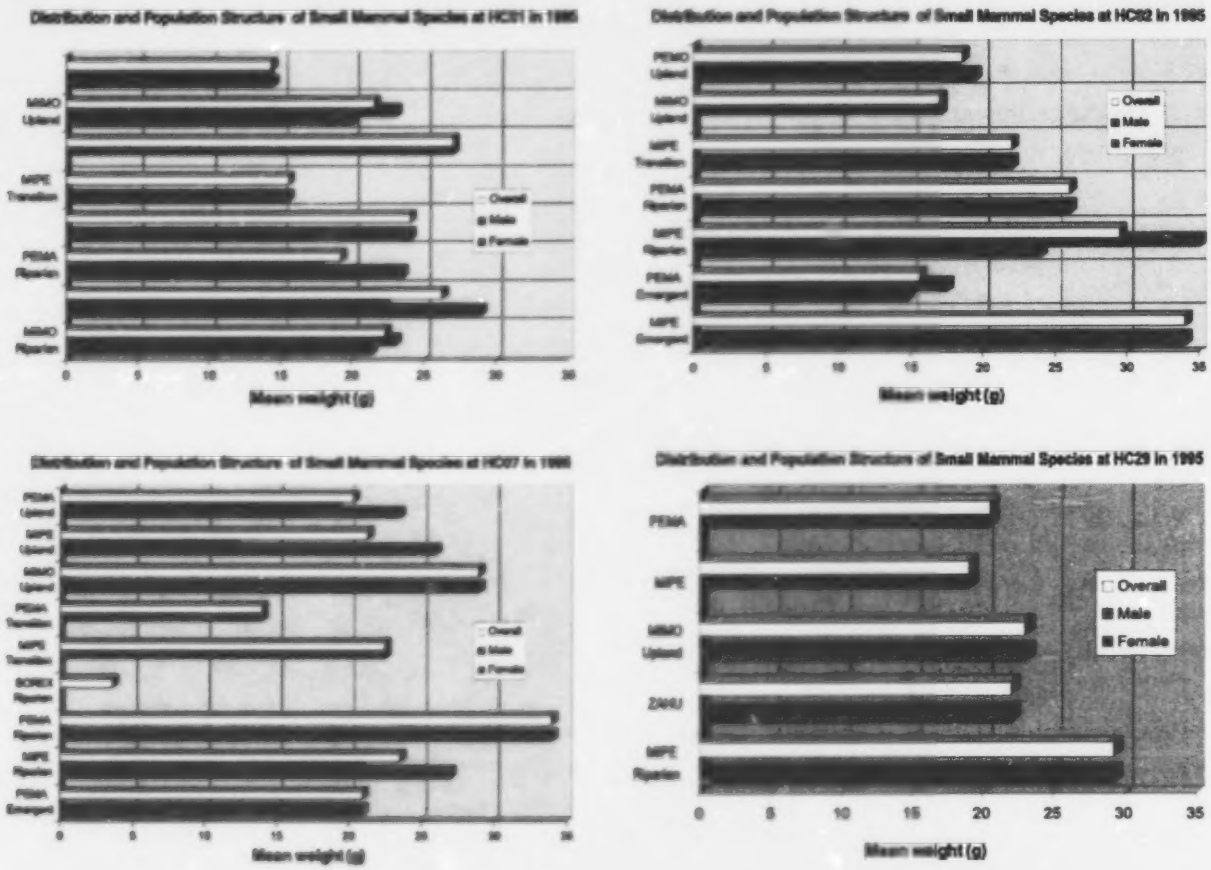


Figure 6a–d. Small mammal species distribution and population structure observed at (a) HC01, (b) HC02, (c) HC07 and (d) HC29 in 1995.

difference in CWD abundance between breeders and non-breeders was recorded only at HC07.

Herpetofauna Community Inventory

Pacific tree frog, western toad, spadefoot toad, long-toed salamander, northwestern salamander, western terrestrial garter snake, and common garter snake were documented (scientific names listed in Appendix 1). Auditory surveys indicated that spadefoot toads were present at HC01, HC04, and most frequently at HC07 and HC10.

DISCUSSION

The results suggest that a riparian shrub component was likely indigenous at most ponds in Hamilton Commonage. At all study ponds the shrub component was either absent or appeared to be threatened by lack of or reduced regeneration (Figs. 5a–d). Ponds with well-developed shrub layers had higher diversity of bird and small mammal species.

The results and observations strongly suggest that historical grazing practices did not sustain the shrub layer in the riparian ecosystem and current practices are not rehabilitating or sustaining woody riparian species. Elmore (1992) reported that most grazing systems are designed to meet the physiological requirements of upland herbaceous plants. These grazing systems can be compatible with riparian herbaceous species, but are not sustainable for woody riparian plants (Elmore 1992). In July and August, when upland forage species have seeded and dried, cattle are attracted to riparian areas where herbaceous vegetation is palatable and nutritious (Skovlin 1984). Cattle will also utilize buds on woody plants, consuming the current year's growth when herbaceous utilization is at 40% (Elmore 1992). Persistence of cattle in the riparian area will result in second and third years' growth removal from shrubs. In independent studies, Schulz and Leininger (1990) and Marcuson (1977) found that shrub production was significantly higher in ungrazed treatments compared with grazed treatments. Knopf and Cannon (1982) also found that size, shape, and volume of willow plants were changed by grazing.

The structural diversity of riparian habitat has been documented as the critical feature that supports wildlife diversity in riparian ecosystems (Thomas et al. 1979). The vertical complexity of riparian ecosystems is created by shrubs and trees, for the most part. At Hamilton Commonage, HC07 had the highest diversity of avian species and the highest structural diversity of aspen-shrub complexes. Bird species present at HC07 and at other ponds with similar aspen-shrub components were not detected in areas or at ponds that lacked a willow component. In particular, some of the species included warblers, flycatchers, warbling vireo, western wood-pewee, and house wren. Kauffman and Kreuger (1984) reported that grazing altered bird species composition and foraging guilds, although bird densities were not affected. The pretreatment inventory results are consistent with results documented elsewhere. The aspen patch at HC07 had the greatest diversity of bird species compared with other study ponds and habitats, and in particular, consistently had the highest incidence of insectivorous and herbivorous/granivorous species guilds. Ponds such as HC01 had large numbers of individual birds, but species composition was lower and composed more of open-ground birds and generalist species, such as shorebirds, vesper sparrows, savannah sparrows, and horned larks. High numbers of waterfowl contributed to the numbers of individuals reported at HC01. Similar results were reported by Bock et al. (1984) where open-ground foragers and species typical of xeric habitat types occurred in grazed habitats.

Vegetation species indicative of long-term grazing impacts, which provided thick ground cover in the riparian zone and relatively more open habitat in the upland zone, provided some benefits as wildlife habitat. Ground nesters and feeders, such as savannah and vesper sparrows benefit from the abundance of invader grasses and increasing rushes and sedges. In the absence of structural diversity, savannah and vesper sparrows will use Canada thistle as calling posts. The herbaceous vegetation development in the wet soils, such as sloughgrass and purple-leaved willow herb, may provide nesting habitat for shorebirds such as rails, yellowlegs, and phalaropes if the vegetation covers a large enough area and open water is nearby; shorebirds, including killdeer and sandpipers, tend to use more open shore situations for nesting.

The pretreatment inventory results showed that small mammal species diversity was also higher in habitats with willow and other shrub species present. Deer mice were found to be in greater abundance in areas that had received more disturbance. Bock et al. (1984) also found deer mice to increase with grazing; however, original species composition was regained 1 year after grazing. Sites where vegetation species were indicative of past disturbance did provide suitable small mammal habitat. For example, grass stands of mature foxtail barley and redtop provided nesting and security cover for small mammals, and resulted in higher numbers, particularly of meadow voles, in riparian zones than upland zones. Deer mice tended to be more abundant in the marginal, open upland zones. Since mature foxtail barley is not consumed by cattle because it causes throat irritation (USDA Forest Service 1937), it can provide suitable habitat unless it is trampled. The results of vegetation structure, CWD, and bare ground analyses indicated that meadow voles in general, and heavier animals overall, tended to occupy sites with the greatest ground cover that had not been recently disturbed. In general, riparian habitat was preferred over upland habitat.

Species composition was influenced by confounding effects from fire suppression. Bird species indicative of coniferous

forest were detected in species inventories and patches of Douglas-fir trees were present throughout the grassland landscape at Hamilton Commonage. British Columbia Ministry of Forests policy has resulted in fire suppression in grassland ecosystems, where a regular fire cycle may have occurred historically. Livestock grazing has also contributed to suppressing fires by reducing fuel loading. Increasing Douglas-fir encroachment into indigenous grasslands will change the wildlife species composition of the riparian and upland communities. As they age and produce a taller canopy, coniferous trees may also utilize the water that would normally have been taken up by deciduous trees and shrubs that occupied riparian zones indigenously.

The variable impacts of grazing indicated that cattle create localized disturbance in riparian areas, including soil compaction, vegetation removal, and mechanical damage (Kauffman and Krueger1984). The results showed that terrain and surrounding landscape played a significant role in affecting impacts by influencing cattle behaviour. Skovlin (1984) reported that cattle behaviour and response to environmental features, such as slope, will affect localized impacts. The grazing systems employed in the pastures where the study ponds are located do not include provision for localized impacts and particularly are not targeted for riparian ecosystems.

Grazing systems have met controversial reviews (Kauffman and Krueger 1984). The pretreatment inventory suggests that terrain and landscape features should be considered in delineating grazing prescriptions. The ponds that received the fewest impacts from grazing were HC07 and HC29, located in the Douglas Lake allotment of Hamilton Commonage. The Douglas Lake pasture received continuous, season-long grazing by horses and cattle. The pasture was large, and contained several patches of deciduous, coniferous, and deciduous-coniferous mixed stands, which provided livestock with plenty of thermal regulating opportunities during the dry, hot summer period. Consequently, localized high impacts that occurred at water sources, primarily from trampling and mechanical damage to important shrub and aspen stands, were reduced by abundant availability of thermal cover. Terrain and landscape features contributed to disproportionate utilization of particular deciduous patches or habitat types.

In conjunction with pasture size, grazing schedules should be included in delineating grazing prescriptions. In the Guichon allotment at Hamilton Commonage, the early rotation of cattle through pastures (in May and early June) likely had far less impact on willow and aspen growth since upland forage was still palatable and temperatures were not yet extreme. However, from mid-June until August, the shrub and aspen components in the Guichon allotment would have received considerable impact. The pastures were smaller than the single large Douglas Lake study pasture and the surrounding landscape within any 1 pasture had far less cover, increasing cattle attraction to the few available aspen-shrub patches. Overall, however, the results strongly suggest trends of unsustainability for willow and aspen components at all study ponds, although rates of change may differ among sites (Figures 5a-d).

As Elmore (1992) has indicated, the physiology of the plants in the pasture must be considered in deriving grazing regimes, and management should follow a prescription procedure that results in a sustainable grazing system for a site. In considering wildlife resources, for example, timing of grazing can be critical. For example, for 2 consecutive years a pair of American avocets were observed to attempt nesting at HC01 on the opposite shore, out of the Guichon pasture. In both years, cattle were introduced into the pasture just as nesting had begun, and within a day all evidence of the nest was gone. Trampling is a critical problem for vegetation and wildlife, and can be avoided for both with appropriate timing of grazing.

SUMMARY AND CONCLUSIONS

The first phase of the long-term grazing impact research program was completed between 1992 and 1996. A research design was developed in the first year and study sites were selected to test the effects of grazing on the biodiversity of grassland riparian ecosystems. In the second year, 1993, sampling trials were initiated at Hamilton Commonage and pretreatment inventories were conducted at several ponds until 1995. Riparian exclosures were installed in 1995 as the experimental treatment.

The multiyear, pretreatment vegetation community results suggest that most graminoid-forb communities were dominated by a few species, which tended to be weedy and therefore indicated varying levels of grazing impact. The restriction of a shrub component to relatively small patches and the lack of evidence for regeneration suggest that long-term, unsustainable trends may be continuing. Impacts may have occurred more slowly at these sites than in others, but the cumulative effects of several years of loss of new growth may result in a significant lack of recruitment and continued loss of the shrub component at a landscape scale.

Although slope was not found to be significant immediately surrounding ponds, observations and results indicated that terrain influenced livestock movement at both a localized and landscape level. Terrain also influenced soil moisture so that moist soils typically occurred on slopes favourable to livestock access, thereby compounding the effects of localized impact. Water accessibility was the primary cause for localized, high impacts at shorelines of moderate terrain with firm soils. Trailing was extensive in these riparian areas, which over the long-term has likely resulted in significant soil compaction that altered the structure and composition of vegetation species and narrowed the riparian

zone. Terrain and soil moisture also influenced the impact to shrub-aspen complexes. Impacts from trampling, grazing, and rubbing were lower when shrub-aspen complexes were not adjacent to water and were situated in a mosaic landscape with several other treed patches.

The consequences of livestock impacts to riparian shrub and tree components can be significant for wildlife species. The results strongly suggest that wildlife species diversity is more dependent on riparian vegetation structure than species. Reduced shrub and deciduous tree components were correlated with lower numbers of wildlife species, although numbers of individuals were not necessarily reduced. Several bird species occurred only in shrub-aspen complexes and appeared to depend on these riparian habitat types to meet basic life requisites.

In general, riparian habitat was used disproportionately to upland habitat by most species of birds and small mammals for several life requisites. Overall, bare ground was associated with generalist species that were typically associated with xeric, upland conditions. Less impact in riparian areas resulted in heavier small mammals, which would have provided higher prey biomass for predator species.

The results suggest that the grazing systems utilized at Hamilton Commonage are incompatible with sustainable management of the woody species component of riparian habitat. For long-term sustainability, management strategies should switch to prescriptions from broad pasture system regimes. Management prescriptions will consider plant physiology, terrain, and integrated resource management objectives.

ACKNOWLEDGEMENTS

Funding for this project was provided by the Habitat Conservation Trust Fund, Interior Wetlands Program (Canadian Wildlife Service and Ducks Unlimited), and Green Plan for Agriculture (Agriculture Canada and B.C. Ministry of Agriculture, Fisheries and Food). D. Low, British Columbia Ministry of Environment, Lands and Parks, initiated this project.

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Appendix A. Common and scientific names of species documented at Hamilton Commonage.

VEGETATION	
Trees	
Pinus contorta	Lodgepole pine
Psuedotsuga menziesii	Douglas-fir
Populus tremuloides	trembling aspen
Shrubs	
Chrysothamnus nauseosus	common rabbitbrush
Salix bebbiana	Bebb's willow
Salix scouleriana	Scouler's willow
Rosa acicularis	prickly rose
Rosa nutkana	Nootka rose
Rubus idaeus	red raspberry
Ribes cereum	squaw current
Forbs	
Achiles millefolium	common yarrow
Agoseris glauca	false dandelion
Allium cernuum	nodding onion
Antennaria dimorpha	low pussytoes
Antennaria microphyla	rosy pussytoes
Artemisia frigida	prairie sagebrush (pasture sage)
Astragalus miser	timber milk vetch
Aster pansus	tufted white prairie aster
Campanula rotundifolia	common harebell
Capsella bursa-pastoris	shepherd's purse
Centaurea diffusa	diffuse knapweed
Centaurea jacea	spotted knapweed
Chamomilla suaveolens	pineappleweed
Chenopodium album	lambs-quarters
Chenopodium capitatum	strawberry-blite goosefoot
Cichorium intibus	common chickory
Cirsium arvense	Canada thistle
Cirsium undulatum	wavy-leaved thistle
Collomia grandiflora	large flowered collomia
Delphinium nuttallianum	wild delphinium
Descurainia pinnata	western tansymustard
Descurainia sophia	flixweed
Epilobium angustifolium	fireweed
Erigeron compositum	cut-leaved fleabane
Erigeron corymbosus	long-leaved fleabane
Erigeron linearis	fine-leaved fleabane
Erigeron filifolius	thread-leaved fleabane
Erigeron flagellaris	wild daisy
Erigeron phyladelphicus	Philadelphia fleabane
Erigeron pumilus	shaggy fleabane
Erigonum heracleoides	parsnip flowered umbrella plant
Erigonum umbrellatum	sulphur flowered umbrella plant
Gaillardia aristata	brown-eyed Susan
Galium boreale	northern bedstraw
Geranium viscosissimum	sticky purple crane's bill
Geum triflorum	purple avens
Grindelia squarrosa	Curlycup gumweed
Lactuca serriola	prickly lettuce
Lepidium densiflorum	common peppergrass
Lithospermum ruderdale	Columbia gromwell, pucoon
Lupinus laxiflorus	lupine (looseflower)
Lotus denticulatis	deervetch
Lupinus sericeus	silky lupine
Medicago lupulina	black medic
Melilotus alba	white sweet clover
Melilotus officinalis	yellow sweet clover
Microsteris gracilis	pink microsteris
Mentha arvensis	field mint
Myosotis laxa	small leaved forget-me-not
Orthocarpus barbatus	bearded owl's clover
Oxytropis campestris	field locoweed
Phacelia hastata	silverleaf phacelia
Phacelia linearis	thread-leaved phacelia
Plantago major	greater plaintain
Polygonum amphibium	water smart weed
Polygonum douglasii	Douglas knotweed
Potentilla anserina	common silverweed
Potentilla gracilis	graceful cinquefoil
Ranunculus cymbalaria	shore buttercup
Ranunculus sceleratus	celery-leaved buttercup
Rhinanthus minor	yellow rattle
Rumex occidentalis	western dock
Salicornia rubra	red European glasswort
Silene douglasii	Douglas catch fly
Smilacina stellata	star flowered Solomon's seal
Sonchus arvensis	perennial sow-thistle
Taraxacum officinale	dandelion
Tragopogon dubius	yellow salsify, oysterplant, goat's beard
Trifolium repens	white clover
Typha latifolia	common cattail
Verbascum thapsus	great mullein
Vicia americana	American vetch
Viola adunca	early blue violet
Zigadenus venenosus	death-camas
Graminoids	
Alopecurus aequalis	little meadow foxtail
Agropyron trachycaulum	slender wheatgrass
Agropyron spicatum	bluebunch wheatgrass
Beckmannia syzigachne	slough grass
Bromus inermis	smooth or Hungarian brome
Bromus japonicus	Japanese brome or cheatgrass
Bromus tectorum	drooping brome grass
Calamagrostis canadensis	bluejoint reedgrass
Calamagrostis rubescens	pine grass
Cerrastium arvense	chickweed
Danthonia intermedia	timber oatgrass
Festuca scabrella	rough fescue
Hordeum jubatum	foxtail barley
Koeleria macrantha	junegrass
Medicago satira	alfalfa
Muhlenbergia asperifolia	scratchgrass
Phleum pratense	common timothy
Poa pratensis	Kentucky bluegrass
Poa sandbergii	Sandberg's bluegrass

Continued...

Appendix 1. Continued.

Puccinellia nuttalliana	Nuttall's alkaligrass
Setaria viridis	green foxtail
Spartina gracilis	alkali cordgrass
Stipa comata	needle-and-thread
Stipa occidentalis	stiff needlegrass
Stipa richardsonii	spreading needlegrass
Stipa spartea	porcupine grass
Rushes	
Juncus balticus	wire rush
Scirpus spp.	bulrush
Triglochin maritima	seaside arrowgrass
Sedges	
Carex rostrata	beaked sedge
Carex aquatilis	water sedge
Horsetails	
Equisetum fluviavile	swamp horsetail
SMALL MAMMALS	
Microtus montanus	Montane vole
Microtus pennsylvanicus	Meadow vole
Peromyscus maniculatus	Deer mouse
Sorex spp.	unknown shrew species
Zapus hudsonius	Meadow jumping mouse
BIRDS	
Actitus macularia	Spotted sandpiper
Agelaius phoeniceus	Red-winged blackbird
Aix sponsa	Wood duck
Anas acuta	Northern pintail
Anas americana	American wigeon
Anas clypeata	Northern shoveler
Anas crecca	Green-winged teal
Anas cyanoptera	Cinnamon teal
Anas discors	Blue-winged teal
Anas platyrhynchos	Mallard
Anas strepera	Gadwall
Aythya affinis	Lesser scaup
Aythya americana	Redhead
Aythya marila	Greater scaup
Aythya valisineria	Canvasback
Branta canadensis	Canada goose
Bubo virginianus	Great horned owl
Bucephala albeola	Bufflehead
Bucephala islandica	Barrow's goldeneye
Buteo jmaicensis	Red-tailed hawk
Catharus ustulatus	Swainson's thrush
Charadrius vociferus	Killdeer
Chlidonias niger	Black tern
Chordeiles minor	Common nighthawk
Cistothorus palustris	Marsh wren
Colaptes auratus	Northern flicker
Contopus sordidulus	Western wood-pee wee
Corvus brachyrhynchos	American crow
Corvus corax	Common raven
Dendroica coronata	Yellow-rumped warbler
Dendroica petechia	Yellow warbler
Dryocopus pileatus	Pileated woodpecker
Empidonax difficilis	Western flycatcher
Eremophila alpestris	Horned lark
Euphagus cyanocephalus	Brewer's blackbird
Falco columbarius	Merlin
Falco sparverius	American kestrel
Fulica americana	American coot
Gallinago gallinago	Common snipe
Geothlypis trichas	Common yellowthroat
Melospiza melodia	Song sparrow
Molothrus ater	Brown-headed cowbird
Numenius americanus	Long-billed curlew
Oxyyura dominica	Ruddy duck
Parus atricapillus	Black-capped chickadee
Parus gambeli	Mountain chickadee
Passerculus sandwichensis	Savannah sparrow
Phalaropus tricolor	Wilson's phalarope
Pica pica	Black-billed magpie
Podiceps auritus	Horned grebe
Podiceps grisegena	Red-necked grebe
Podiceps nigricollis	Eared grebe
Podilymbus podiceps	Pied-billed grebe
Pooecetes gramineus	Vesper sparrow
Porzana carolina	Sora
Rallus limicola	Virginia rail
Recurvirostra americana	American avocet
Regulus calendula	Ruby-crowned kinglet
Sialia currucoides	Mountain bluebird
Sitta canadensis	Red-breasted nuthatch
Spizella passerina	Chipping sparrow
Sturnella neglecta	Western meadowlark
Sturnus vulgaris	European starling
Tachycineta bicolor	Tree swallow
Tachycineta thalassina	Violet green swallow
Tringa melanoleuca	Greater yellowlegs
Turdus migratorius	American robin
Tyrannus tyrannus	Eastern kingbird
Wilsonia pusilla	Wilson's warbler
Xanthocephalus xanthocephalus	Yellow-headed blackbird
HERPETOFAUNA	
Ambystoma gracile	Northwestern salamander
Ambystoma macrodactylum	Long-toed salamander
Bufo boreas	Western toad
Hyla regilla	Pacific tree frog
Scaphiopus intermontanus	Spadefoot toad
Thamnopis elegans	Western terrestrial garter snake
Thamnopis sirtalis	Common garter snake

Removal of the Riparian Zone During Forest Harvesting Increases Stream Temperature: Are the Effects Cumulative Downstream?

J. M. Shrimpton
Biology Program, University of Northern British Columbia
3333 University Way, Prince George, BC, V2N 4Z9, Canada
shrimptm@unbc.ca

J. F. Bourgeois
Biology Program
University of Northern British Columbia,
3333 University Way
Prince George, BC, V2N 4Z9, Canada

J. T. Quigley
Faculty of Forestry
University of British Columbia
and Habitat and Enhancement Branch
Fisheries and Oceans Canada
360 – 555 West Hastings Street
Vancouver, BC, V6B 5G3, Canada

D. M. Blouw
Research and Graduate Studies
University of Northern British Columbia
3333 University Way
Prince George, BC, V2N 4Z9, Canada

ABSTRACT

The experimental removal of riparian vegetation was performed to determine the impact on stream temperature. Water that flows through the unbuffered creeks heats more than the water in the buffered creeks. As a consequence, the diurnal change in temperature is greater in the unbuffered creeks. We do not know many of the moderating forces on heat loading that may be at work in northern Interior streams. As a result we have heated selected streams to uncouple changes in temperature from other aspects of the natural environment that often co-vary with temperature. Energy put in as heat is transmitted a considerable distance downstream. The loss of heat after increasing stream temperature at a single point, however, appears to be independent of riparian zone characteristics.

Key words: forest harvesting, riparian vegetation, stream temperature.

Water temperature is an important parameter influencing stream ecology and fish productivity. Stream temperature plays a central role in the aquatic ecosystem because it influences all biological processes including metabolic rates, growth, behaviour, and survival of fish populations. There is a narrow range of temperatures over which salmonids can thrive. At the upper and lower thresholds of this optimum range, fish exhibit behavioural and physiological responses that enable them to maintain homeostasis. Fluctuations in daily temperature are also stressful for fish and impact on energy reserves (Thomas et al. 1986).

Clearcut logging has been directly related to increases in stream temperature in a number of studies (Ringler and Hall 1975, Rishell et al. 1982, Lynch et al. 1984, Holtby 1988). Leaving an undisturbed buffer strip of riparian vegetation adjacent to the stream has been shown to mitigate the increase in stream temperature (Newbold et al. 1980, Barton et al. 1985). The importance of riparian reserves is reflected in harvest prescriptions mandated by the British Columbia Forest Practices Code Act (FPC). The FPC defines the dimensions of buffer strips that must be left adjacent to streams dependent on physical and biological attributes of the streams. For small stream reaches <1.5 m wide, however, the FPC prescribes a 20-m riparian management zone, which allows for a wide range of practices within the riparian zone of the stream, including removal of all timber. Our first objective has been to determine the value of these buffer strips <30 m in width for moderating daily fluctuations in temperature within these small streams in central British Columbia.

Our second objective has been to evaluate the transmission of heat downstream. Due to the high latent heat capacity of water, changes in temperature may be efficiently transmitted downstream. The effect of warming many small streams may, therefore, be incremental, such that the potential cumulative downstream impact may be significant. To evaluate the efficiency of heat transmission downstream, we are manipulating stream temperatures. This approach allows us to uncouple changes in temperature from other

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

aspects of the natural environment that often co-vary with temperature (for example discharge, cloud cover, precipitation, air temperature, insolation, groundwater flows). With the wide fluctuations in temperature that we observe, this experimental approach enables us to clearly isolate changes in temperature attributable to harvest practices and their downstream effects. A second major benefit to experimentally manipulating temperature is that we will be able to quantify the fate of the heat loading that is well documented to take place in water courses adjacent to clearcuts. The downstream transmission of upstream temperature events has not been investigated. They may have considerable importance in the incremental impact of harvesting numerous small cutblocks throughout a drainage basin, or throughout an entire watershed.

METHODS

Assessment of Riparian Reserve Zones

We have been examining environmental changes that occur due to forest harvest practices to evaluate the effectiveness of the FPC in moderating stream temperature variation. In the Torpy River watershed, 150 km east of Prince George, B.C., a cutblock that had a Y-shaped creek (referred to as "Y Creek") flowing through the centre of the block was harvested in January 1996. The 2 forks of the creek have an average width of <1.5 m and discharge of approximately 100 L/min in each fork. A 10-m riparian reserve zone containing deciduous trees and coniferous regeneration trees was left adjacent to both forks of the creek. Over a distance of 350 m on the north fork of Y Creek, we had this riparian reserve zone removed. The south fork reserve was left intact. The 2 forks of the creek are very similar in slope aspect, discharge, and understory vegetation. To observe changes in temperature in Y Creek, we are using remote, self-contained temperature data loggers (Onset Instruments, Cambridge, MA) programmed to record temperature hourly. In the north fork we placed temperature loggers in the stream where it entered the cutblock, and at the top, mid-point, and bottom of the experimental section where the riparian reserve zone was removed. Temperature loggers were placed at similar points in the control south fork. Additionally, temperature was recorded below the confluence.

Instream Heat Manipulation

For the temperature manipulation experiments, water was pumped from a stream using a gasoline-powered, variable-speed Honda portable pump (Model WB15) connected to a noncollapsible hose. The intake consisted of a filter basket covered by fine netting to prevent damage to the pump by sediment. Water was then pumped through propane-powered, instantaneous water heaters (Bosch Booster Pressure Wash Model W400K5, 117,000 btu) and reintroduced into the stream through a diffuser system to ensure mixing. Each diffuser unit was constructed from 5 household showerheads in a manifold-type configuration.

Temperature loggers were deployed in a stream several days before and after the temperature manipulation was scheduled to take place. Two temperature loggers were always placed a few metres upstream from the heating apparatus and the rest were placed downstream at regular intervals in the test stream and in any tributaries present. The loggers were programmed to measure and record temperature every 5 minutes. This enabled us to calculate the background variations in temperature observed in the test streams. Establishing the background conditions was essential in order to quantify the transfer of heated water downstream. The difference between the observed temperatures during the heating experiments and those calculated based on the natural temperature regime of the stream is expressed as ΔH. With this approach we could examine physical factors that affect the rate of heat dissipation following temperature manipulation.

RESULTS

Y Creek

A summary of temperature changes for 1 day in June of 1997 is shown in Figure 1. Water temperatures in the forested areas above the clearcut were very similar in the north (stn. 19) and south (stn. 20) forks of Y Creek, and there was relatively little diurnal change in temperature. The north fork (stn. 6) and south fork (stn. 5) had comparable water temperatures before they entered the experimental area. Water flowing out of the unbuffered north fork (stn. 3) heated more than water flowing out of the buffered south fork (stn. 4). As a result, the diurnal difference in temperature was greater in the unbuffered fork. There was an increase in variance in water temperature that occurred in unbuffered sections of stream relative to forested areas or streams with riparian buffer strips. Water below the confluence (stn. 17) is a rough average of the 2 inflowing streams.

Throughout the early part of the year, the average temperature differential (ΔT) between the upper and lower stations on each fork of the creek was positive (Table 1), but it was greater and showed greater variance on the unbuffered fork. During the fall, ΔT became negative for the unbuffered fork, indicating a decrease in stream temperature as the water flowed through the experimental section. This loss in water temperature was not seen to the same degree for the stream with a 10-m riparian reserve zone.

Instream Temperature Manipulations

An example of data from a heating experiment is shown in Figure 2. Hooker's Stream is a north-side S4 tributary of the Lower Torpy River that flows through a cutblock. The

Table 1. Mean temperature difference (ΔT) from 1300 hr to 1800 hr for April–November, 1997. ΔT-Unbuffered is calculated from the difference between temperature stations 3 and 6 on the experimental sections of Y Creek; ΔT-Buffered is calculated from the difference between stations 4 and 5 on the control sections of Y Creek. Distance between stations is 350 m. Station 17 is 50 m below the confluence of the 2 forks of Y Creek. Air temperature is averaged over the month for temperatures recorded between 1300 hr and 1800 hr.

Month	Mean temperature difference (°C)		Stn. 17 temperature	Air temperature
(1997)	ΔT-Unbuffered	ΔT-Buffered	(°C)	(°C)
Apr	-0.19	-0.15	0.35	8.2
May	0.66	0.41	5.76	15.5
Jun	0.73	0.25	9.54	17.4
Jul	0.63	0.31	10.28	19.8
Aug	0.58	0.21	11.74	21.7
Sep	0.24	0.16	9.39	17.3
Oct	-0.01	0.14	4.86	5.0
Nov	-0.09	-0.02	0.44	-1.2

discharge was approximately 150 L/min. A heating experiment was conducted on 8 October 1997 between 1210 and 1740 hr. The heating apparatus was located 158 m upstream from the lower block boundary. This boundary is the edge of the riparian zone along the north bank of the Torpy River. Figure 2 indicates the location of the block boundary, as well as the locations of a deactivated road and 2 tributaries flowing into the test stream.

Figure 2 supports the assumption that energy put into the stream as heat is transmitted a considerable distance downstream. The linear negative slope (-0.0061) of the data indicates that heat from the stream is dissipated at a constant rate of 0.0061°C/m. During a second heating experiment in the morning on 9 October, the heat loading dissipated at a rate of 0.0058°C/m. The slopes of these 2 lines do not differ significantly and there is little difference in dissipation rates between the 2 experiments. There was only a 5°C difference between mean air temperature during both experiments and

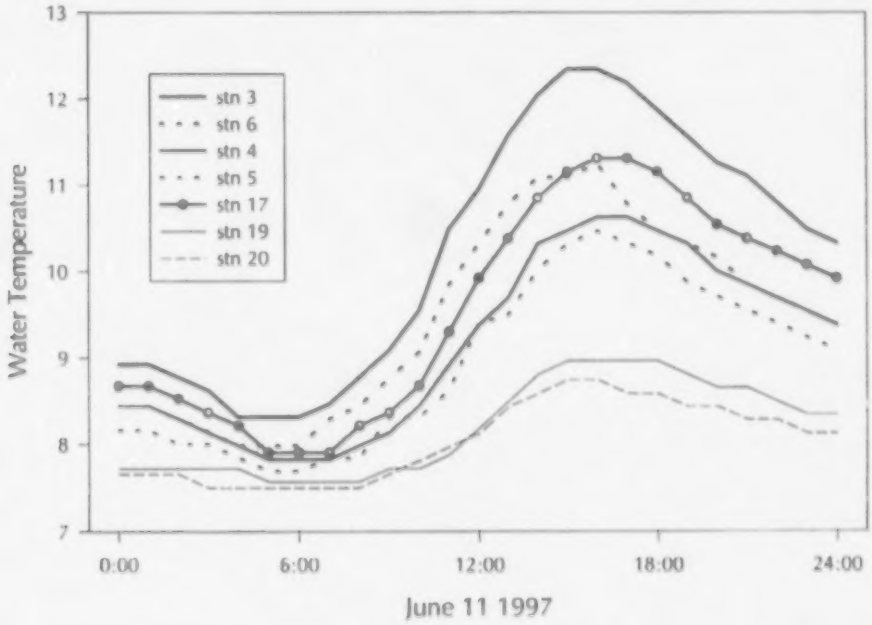

Figure 1. Daily change in water temperature in Y Creek on 11 June 1997. Locations of the temperature recording stations are described in the text.

no measurable rainfall during either of the heating experiments on Hooker's Stream. In additional experiments we found air temperature to have more of an effect on the dissipation of heat.

Heat dissipation was linear with distance from the heat source. As the stream flow passed across roads, the temperature signal was more variable. The overall net rate, however, did not change appreciably. Figure 2 also shows the rate of heat dissipation for the stream flowing through a cutblock and through an unharvested reach. The loss of heat from the stream continues at a constant rate. There appears to be no difference between the rate of heat loss through the cut block or the reaches with riparian vegetation adjacent to the stream.

DISCUSSION

Water temperature of streams increases following clearcut logging. The principal cause of elevated water temperature following logging has been attributed to increased surface exposure due to direct insolation (Brown and Krygier 1970, Sinokrot and Stefan 1993). The retention of buffer strips adjacent to streams has been reported to be effective in limiting increases in stream temperature attributable to forest harvesting (Hewlett and Fortson 1982, Barton et al. 1985), and riparian buffer strips of 30 m were found to provide sufficient shading and prevent increases in stream temperature. In British Columbia, the FPC does not require buffer strips of this size to be retained on small streams <1.5 m wide. The presence of a 10-m riparian buffer strip, however, does moderate increases in stream temperature. The removal of this 10-m riparian reserve strip resulted in substantial differences in heat gain between the forks of Y Creek (Table 1).

Diel fluctuations in stream temperature also increase following clearcut harvesting (Lynch et al. 1984). Retention of riparian reserves adjacent to streams limits diel fluctuations in temperature. This is shown by the diel fluctuations in temperature that were greater in the unbuffered stream than in the buffered stream (Fig. 1). Although the daily fluctuations in stream temperature are less due to the presence of a riparian buffer strip, they were still much greater than temperature oscillations observed in both streams prior to entering the cutblock. Above the cutblock, both streams showed mean daily temperature oscillations of <1.0°C. Heating of streams with riparian buffer strips of this width, therefore, will still occur.

Following clearcut harvesting within the Carnation Creek watershed on Vancouver Island, Holtby (1988) reported an increase in mean stream temperature for all months of the year. There was variation over the year in the magnitude of the increase in water temperature, which was greatest in August and smallest in December. In contrast, we found that removal of the riparian zone resulted in increases in water temperature (ΔT positive) only during the summer months. In April, October, and November, water flowing through the

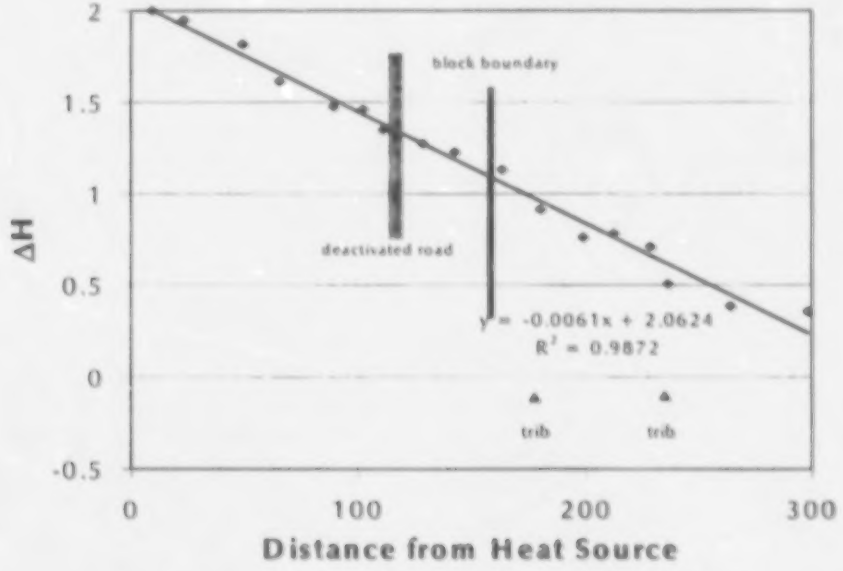

Figure 2. Results of a heating experiment conducted on an S4 stream on the north side of the Torpy River. All represent the difference between the observed temperatures during the heating experiment and what was expected based on the natural changes in water temperature.

unbuffered experimental section of the stream lost heat and decreased in temperature (Table 1). A loss of heat was also seen in the stream with a buffer strip during these months, but ΔT was smaller. The loss of heat from streams during fall and winter months has also been reported for streams flowing through harvested area in Pennsylvania (Rishel et al. 1982). Ambient air temperature is an important factor in heat gain and loss in these small streams and likely accounts for these findings.

The difference in heat gain between the protected and unprotected forks was easily measurable over the short distance involved (350 m). Presumably, the increase in temperature over longer unbuffered distances will be greater. The continued transmission of heat downstream is reflected in the water temperature of station 17 (Fig. 1). Water temperature below the confluence of the 2 forks of Y Creek reflects blended water from the experimental and control reaches. The fact that the temperature was a rough average of temperatures in the 2 forks of the stream suggests that heating of water within the clearcut is transmitted downstream with considerable efficiency. Temperature gains on a watershed level following forest harvesting have been documented. Ringler and Hall (1975) found the mean water temperature of streams in Oregon increased with extent of the watershed clearcut. The extent of harvest within a watershed, therefore, will affect the magnitude of temperature changes. Incremental increases in water temperature from many clearcuts may result in a cumulative increase in water temperature.

The rate of heat loss is linear over distance (Fig. 2). The decrease in ΔT with increase in distance from the heat source in our heating experiments is a result of heat loss due to long-wave radiation, heat loss due to evaporation of water from the water surface, convection of heat across the air-water interface, and conduction of heat to the streambed (Brown 1969, Sinokrot and Stefan 1993). In heat loading of streams, thermal radiation has been shown to be the dominant source of energy, but other components are not negligible. Radiation of heat from the stream to the atmosphere, therefore, may be the most important component of heat flux to cool the stream. If this is the case, we would not expect forest canopy to affect the rate of heat loss. The constant rate of heat loss within the clearcut and within the unharvested reach indicate that riparian vegetation has little influence on dissipation of heat. Other factors, therefore, affect the rate of heat loss. Air temperature will affect the evaporative flux and convective flux of heat. Temperature manipulation experiments conducted at night when air temperature is lower result in higher rates of heat loss from the streams. Increases in wind speed will also affect the convective flux and evaporative flux of heat from the stream. Streambed substrate was predominantly gravel, which has been found to have low thermal conductivity (Brown 1969).

Changes in stream temperature following timber removal have been documented in our research. The downstream transmission of upstream temperature events has considerable importance to cumulative heat loading following harvesting of numerous small cutblocks throughout a drainage basin, or throughout an entire watershed. As harvest practices cause incremental heat loading, considerable impacts may occur with respect to the cold-adapted salmonids that are the mainstay of the British Columbia fishing economy.

ACKNOWLEDGEMENTS

This research was supported by grants to DMB from Forest Renewal British Columbia and the British Columbia Ministry of Environment, Lands and Parks.

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Wetland Classification and Habitats at Risk in British Columbia

Will MacKenzie
Research Branch, British Columbia Ministry of Forests
Bag 5000, Smithers, BC, V0J 2N0 Canada
Will.MacKenzie@gems1.gov.bc.ca

Jennifer Shaw
Ecological Consultant
Box 2463, Smithers, BC, V0J 2N0 Canada

ABSTRACT

The British Columbia Ministry of Forests is developing a practical classification system for British Columbia's wetland and riparian ecosystems. The project has identified a diversity of wetland and riparian ecosystem types, in terms of environmental profiles, regional distribution, and habitat values. Urbanization, agricultural development, hydrological modification, livestock grazing, intensive forestry, and introduction of invasive species pose the gravest threats to wetland and riparian ecosystems by outright destruction, habitat simplification, or isolation from important adjacent habitats. However, some wetland types are more sensitive to these disturbances or provide greater habitat values than others. The classification recognizes this variability in ecosystem function and can be used to identify and quantify ecosystem types of greatest concern. Given the importance of wetland and riparian ecosystems to overall ecosystem integrity, a wetland management policy that relies on the best available scientific information to protect these habitats is crucial. This classification project provides one such body of scientific knowledge that could form the foundation of a site- or region-specific policy for wetland and riparian conservation and management.

Key words: classification, ecology, hydrogeomorphic, riparian, wetlands.

Wetlands and riparian areas provide wildlife, fisheries, biodiversity, water quality, and aesthetic values that are disproportionately large compared with their limited extent in the landscape (Forman and Godron 1981, Naiman and Decamps 1990, Pinay et al. 1990, Gregory et al. 1991, Malanson 1993, Mitsch and Gosselink 1993). Recognition of the special nature of these habitats by researchers and managers of natural resources has led to conservation legislation and regulations in many jurisdictions. In British Columbia, the Forest Practices Code provides special management protection of riparian and wetland ecosystems. Guidelines for riparian management in the *Riparian Management Area Guidebook* (RMAG) (B.C. Ministry of Forests and BC Environment 1995) use an administratively and operationally simple classification system. However, the RMAG does little to address ecological differences between specific wetland and riparian types. Recognition of these differences is required to achieve "best management practice" of wetland and riparian areas or to make informed interpretations for other tasks such as risk ranking or wildlife habitat evaluation.

An ecologically based classification system is one of the most important tools for understanding ecosystems and applying ecosystem management principles. Classifications allow for ordering, comparison, synthesis, mapping, and inventory of information and give resource workers a common language to communicate results (Lotspeich and Platts 1982). Biogeoclimatic ecosystem classification (BEC) of forested sites is well developed in British Columbia (Pojar et al. 1987) and is used extensively in many aspects of natural resource management. To date, this classification has been applied primarily to forested ecosystems of the province. To fill the information gap surrounding wetlands and non-forested riparian ecosystems, British Columbia's Ministries of Forests, and Environment, Lands and Parks, through Forest Renewal BC funding, began the Wetland and Riparian Ecosystem Classification (WREC) project.

British Columbia has a high diversity of wetland ecosystems and landscapes. The WREC project has described >100 distinct wetland and related ecosystems, each with unique ecological functions and distributions. Some of these wetland and "riparian" ecosystems may be at risk in British Columbia. The WREC project will provide tools for recognizing, evaluating, and mitigating risk to these ecosystems. The following discussion outlines the range of risks to wetlands, the major attributes of the classification, some observed risks to wetlands by wetland regions, and the ways that the classification could be applied to risk assessment and management.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

WETLANDS AS HABITATS AT RISK

Wetland and flood ecosystems have high wildlife values particularly in arid climates where the distinction from upland ecosystems is most pronounced. These habitats are vital for wetland-dependent species, such as amphibians and water birds, and important for upland species that use wetlands and their associated riparian areas for food, water, and cover. Some features that may influence a wetland's wildlife habitat value are water, structural diversity and cover, abundant forage, high prey densities, and unique habitat. Land management activities that affect these attributes will impact the use of the site by wildlife.

What Is Risk?

The Concise Oxford Dictionary (1995) defines risk as a chance or possibility of danger, loss, injury, or other adverse consequences. Historically, wetlands have been exposed to high levels of risk due to poor public perception and valuation. Since early European settlement, the perception of wetlands as wastelands, available for conversion to more productive uses, has caused millions of hectares of wetlands across Canada to be drained or filled (Lands Directorate 1986). For this paper, we address risk as it applies to wetland and riparian habitats. We define risk to habitat as a function of sensitivity to disturbance, potential for exploitation, and loss of core habitat values. The 2 distinct types of habitat risk are risk of change or destruction (e.g. conversion to agricultural land) and risk of degradation or loss of value (e.g., reduce wildlife habitat capability). Habitat degradation can change ecosystem composition and structure fundamental to wildlife use without complete ecosystem destruction.

Risks to Wetlands

Our discussion will focus on habitat destruction and degradation, habitat fragmentation, and management policy as major contributors of risk.

Habitat Destruction and Degradation

In Canada, wetland area has decreased by 15% since European settlement (Lovett Doust and Lovett Doust 1995). Likewise, human activities such as urban development and resource extraction have resulted in wetland loss and degradation throughout British Columbia.

In general, scientists agree that habitat destruction is currently the primary threat to species diversity worldwide (Ehrlich 1988, Wilson 1992 cited in Wilcove et al. 1998). In British Columbia, wetlands are at risk from hydroelectric initiatives, urban development, agriculture, forestry, ranching, transportation networks, mining, and outdoor recreation.

Hydroelectric development has caused losses of wetland and riparian ecosystems throughout the province. Impoundment of reservoirs as well as downstream impacts can present a serious risk to wetland and riparian ecosystems and the species they support. The most significant impoundment impact is the flooding of valley-bottom ecosystems after dam construction. Large-scale projects such as those on the Columbia and Peace rivers in the 1960s resulted in flooding of huge tracts of valley-bottom ecosystems including important wetland and riparian habitats (Sandborn and Penfold 1996). Although impoundment impacts can be substantial, downstream ecosystems are also at risk (Poff et al. 1997). Fluctuating water levels causing changes in flood regimes can destroy communities such as cottonwood (*Populus balsamifera*) floodplain sites that provide important wildlife habitat (Rood et al. 1995). The search for energy alternatives has introduced new threats to wetland ecosystems including the harnessing of tidal power and the extraction of peat to generate electrical energy (National Wetlands Working Group 1997).

Urban developments can pose a wide range of threats to wetlands. The most obvious threat comes from draining wetlands to accommodate commercial, housing development, and infrastructure (National Wetlands Working Group 1997). In British Columbia, this is especially common in the Fraser Valley and south Okanagan where population pressures are high. Sandborn and Penfold (1996) estimate that 30% of all land converted for urban development between 1967 and 1982 was natural wetlands. Additional impacts associated with urban development such as nutrient loading, introduction of toxic materials, and interruption of local hydrologic patterns can damage or destroy remaining wetlands (Bunnell et al. 1995). These problems are also associated with mining and transportation activities. Two impacts of transportation networks, in-filling and damming, are evident along many of British Columbia's highways.

Agriculture has been a primary cause of wetland and riparian habitat destruction in Canada. The Lands Directorate (1986) claims that 80% of the Fraser River Delta has been lost, largely to agriculture. One such example is the draining and filling of the 11,700 ha Sumas Lake wetland in the 1920s (Sandborn and Penfold 1995). Clearing, however, is not the only threat. Wetlands beside agricultural fields can also experience nutrient loading, reduced water flows, and pesticide accumulation. Wetland conversion of fens, meadows, and swamps to hay fields significantly threatens wetland and riparian habitats in the southern and central Interior. Willow swamps, for example, are often converted to reed canarygrass (*Phalaris arundinacea*) to provide winter forage for livestock. Livestock grazing in riparian areas can limit or exclude use by wildlife thereby decreasing their available habitat. Bunnell et al. (1995) cite Green and Kauffman's (1989) observations of effects of cattle grazing in riparian areas:

- direct vegetation damage caused by browsing or trampling;
- changes in plant communities through selective browsing;

- soil compaction and disturbance that increase erosion and decrease water availability to plants;
- changes in fluvial process, which lower water tables or decrease available sites for invasion of woody species; and
- changes to aquatic ecosystems through sedimentation and fluvial process, as well as changes to water quality due to fecal inputs.

Harvesting of swamp forest and alteration of existing wetlands through road building has reduced wetlands in some areas. Marginally productive swamp forest are commonly drained to improve tree growth in the boreal forest outside British Columbia and may become more important here as timber reserves diminish.

Although outdoor recreation is often associated with habitat protection, certain activities and development practices can destroy wetland and riparian habitats. Destructive uses of wetlands can include developing boating facilities, creating bathing beaches, and using off-road vehicles. In the United States, recreation activities are implicated in the demise of 27% of all endangered species—13% of all endangered species are threatened by off-road vehicles (Wilcove et al. 1998).

Invasive species may also threaten some wetland ecosystems. Shallow open water and marsh communities in southern British Columbia, for example, are threatened by nonnative species such as purple loosestrife (*Lythrum salicaria*) and Eurasian water-milfoil (*Myriophyllum spicatum*). Degraded habitats can become susceptible to invasions of nonnative species, which can displace critical species upon which wildlife depend, thereby reducing habitat value.

Climate change is anticipated to directly impact the range, composition, and viability of ecosystems and individual species alike. Some climate models predict a 0.9–3.5°C increase in global mean temperature over the next century (Houghton et al. 1995 cited in Wilcove et al. 1998). The resulting rise in temperatures and sea levels may drastically affect habitat viability, composition, and distribution. Rapidly increasing sea levels may destroy critical estuary habitats used by migrating waterfowl and other resident animals.

Habitat Fragmentation

The cumulative effects of resource extraction, urban development, and transportation networks have resulted in a disconnected landscape. Noss (1994) cites Shaffer's (1992) assertions that long-term viability of large carnivores in North America is being compromised by the fragmentation of the original wilderness into small refugia. This phenomenon may be occurring at smaller scales for less mobile wetland animals, as wetlands become isolated from each other by development activities and resource extraction. Semlitsch and Bodie (1998) argue that small wetlands are extremely valuable for maintaining diversity in a number of plant, invertebrate, and vertebrate taxa (e.g., amphibians). The continuing disappearance of small wetlands will cause a dire reduction in the ecological connectivity among remaining species populations.

Policy Initiatives

Despite improvements in recent years, wetland and riparian ecosystems continue to be under-protected in Canada. In fact, many policies have actually encouraged wetland conversion. For example, in Ontario, until 1988, wetlands were taxed twice as heavily when left in a natural state than if they were drained and planted in crops (Lovett Doust and Lovett Doust 1995). This example illustrates the need for additional wetland policies that emphasize their ecological value and mitigate the threats discussed here. Often, however, there is an equally strong need to better understand existing policies.

In 1986, the Canadian Fisheries Protection Act imposed a "no net loss of fish spawning habitat" policy. This policy requires developers to create an equivalent area of wetland to that destroyed by the project (Lovett Doust and Lovett Doust 1995). The United States has implemented similar wetland mitigation policies. However, the regulations do not specify what kind of wetland. Therefore, complex wetlands are frequently replaced by less complex types that are easier to establish. This policy does little to deter development (Roberts 1993 cited in Mitsch et al. 1998).

Semlitsch and Bodie (1998) have shown that current and proposed legislation in the United States inadequately maintains regional wetland diversity in amphibians. They further suggest that such legislation is inadequate for some taxa of plants, micro-crustaceans, and insects that use small wetlands. To protect the ecological connectivity and source–sink dynamics of species populations, they strongly advocate that wetland legislation focus not just on size but also on local and regional wetland distribution (Semlitsch and Bodie 1998).

WETLAND CLASSIFICATION

The Wetland and Riparian Ecosystem Classification (WREC) Project

The Wetland and Riparian Ecosystem Classification (WREC) is a British Columbia Ministry of Forests initiative to improve ecological understanding of wetland ecosystems and riparian areas. The project has 3 principal objectives:

1. to develop a wetland classification framework that parallels biogeoclimatic ecological classification (BEC) and recognizes some of the unique hydrological attributes of wetland ecosystems;
2. to classify and describe provincial site associations for wetland and riparian ecosystems and propose management interpretations; and
3. to develop supporting materials such as botanical identification keys and standardized terminology.

The scope of WREC encompasses all true wetlands as well

as related ecosystems (Fig. 1). These include non-forested "riparian" or flood ecosystems, vegetated estuarine ecosystems, and transitional non-wetland ecosystems (i.e., the shrub–carrs and wet meadows described by Runka and Lewis [1981]) that occur with wetlands but do not meet the criteria for wetland designation. This program will produce a complete description of the classification framework (MacKenzie and Banner in prep.), description of provincial wetland site associations, a hydrophyte species list (MacKenzie 1998), and standardized terminology and methods for wetland description.

WREC Field Sampling and Observations on Wetlands at Risk in British Columbia

Extensive wetland sampling following standard field methods (B.C. Ministry of Forests and B.C. Ministry of Environment, Lands and Parks 1998) has been ongoing since 1995 as part of the WREC project. Sampling was primarily on Crown lands of the Interior and North Coast. This data, combined with existing wetland and riparian plots from BEC sampling in the 1970s and 80s, various academic theses, and terrestrial ecosystem mapping projects, sum to approximately 3,000 wetland and riparian ecosystem plots. From this data, WREC has identified approximately 100 distinct wetland, flood, and estuarine site associations. Each of these community types has distinct ecological functions, habitat values, and sensitivities.

Our sampling focused on areas with (relatively) high wetland densities that remain comparatively undisturbed. At the landscape level, wetlands are generally scarce and the number of high density areas is limited.

Our observations suggest that the most prevalent disturbances to wetlands on forested Crown lands are from poor road building and culvert placement. Impoundment of small seeps by roads transforms ecosystems to wetter types, converting swamps to shallow water or marsh habitats. In addition to road building, harvesting in and around wetlands can have obvious impacts. For example, we have noticed small wetlands in clearcuts have dried up after their drainage patterns were altered by vegetation removal and topography alteration. However, whether this is entirely due to harvesting is unclear. Forest harvesting is now restricted around larger wetlands, but small wetlands, which are more likely to experience negative impacts from riparian harvesting, remain unprotected (B.C. Ministry of Forests and B.C. Environment 1995). Impacts to wetlands from riparian harvesting are the topic of many unanswered research questions.

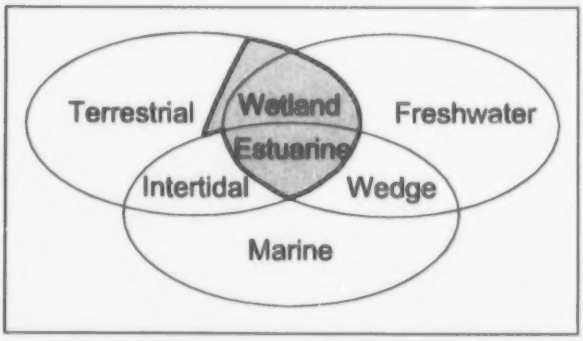

Figure 1. Ecosystem realms addressed by the Wetland and Riaprian Ecosystem Classification (WREC).

The most striking examples of wetland and riparian degradation we encountered during field sampling were on private land. Wetland conversion to agriculture, draining and ditching projects, and overgrazing were all observed. The extent of wetland loss from these activities is unknown. The distribution of these activities, however, suggests they have contributed to the decline in overall wetland area as well as the relative rarity of communities such as Bebb's willow (*Salix bebbiana*)–bluejoint (*Calamagrostis canadensis*) in the Chilcotin and southern Interior.

We recognize several informal wetland regions distinguished by patterns of occurrence in the landscape and the types of wetland and related ecosystems found there. We describe each of these regions (the biogeoclimatic zones they encompass in parentheses) and some risks to the wetlands of each.

Hyperoceanic Lowlands (CWHvh, vm)

This is a 30-km wide band of low rolling terrain along the North Coast. This wetland region experiences a hypermaritime climate characterized by extreme rainfall and moderate annual temperatures. The geology is primarily granite pluton, poor in minerals with little or no glacial till. We estimate that wetlands cover almost one-half of the land base in this area. Wetland types are primarily blanket bogs and bog woodlands of shore pine (*Pinus contorta* var. *contorta*) and yellow-cedar (*Chamaecyparis nootkatensis*) on gently to strongly sloping terrain. These community types and this wetland-dominated landscape are unique in British Columbia and globally. Because of poor growing conditions and difficult access, there has been little pressure to harvest the forested wetlands of the outer coast. Fjord estuaries with extremely high wildlife values are also found in this region. While upland bogs are not generally at risk in this region, North Coast estuaries have experienced relatively high levels of disturbance from log handling activities. More critically, rapid rises in sea levels generated by global warming could destroy a major portion of these estuaries.

Temperate South Coast (CDF, dry CWH)

The southeast corner of Vancouver Island, the Gulf Islands, and part of the adjacent mainland have a Mediterranean climate. Wetlands are somewhat common. The northern range limit of many plant species is within this region and this, combined with the mild climate, produces ecosystems not

found elsewhere in British Columbia. Estuarine ecosystems are distinct from those on the North Coast. Urban and agricultural development pressures are extreme throughout most of this region, and large holdings of private land and strong development pressure have placed many wetlands at risk. Two striking examples are Burns Bog in Burnaby and Rithet's Bog north of Victoria.

Arid Interior (BG, PP, IDF, MS)
The dry Interior of the province is a mix of grasslands and dry forest types. The climate is hot and dry with little growing season precipitation or snowfall. Wetlands are uncommon and are usually marshes or swamps; peatlands occur only at higher elevations (IDF and MS). Saline meadows and some marsh types such as woolly sedge (*Carex lanuginosa*) and spangle-top (*Scolochloa festucacea*) are most common (possibly unique) to these dry regions. Because water sources are fewer and wetlands are markedly different from adjacent upland habitats in structure and species composition, these ecosystems provide critical wildlife habitat. Larger wetlands in this region such as the Columbia marshes have extremely high levels of wildlife use. In addition, certain wetland-dependent wildlife species at risk, such as the Great Basin spadefoot (*Spea intermontana*) and tiger salamander (*Ambystoma tigrinum*), occur in this region. Urban development and agriculture present high threats to wetlands in this region.

Sub-boreal/Boreal Interior (BWBSdk, SBPS, SBS)
Wetlands are relatively common in the sub-boreal and boreal regions. Short cool growing seasons promote formation of peatland in most wet depressions. Consequently, subdued terrain may be covered by large expanses of peatland. Marshes and swamps are commonly associated with lake and river systems. Risks to wetlands are generally few except in areas of agricultural development such as valley bottoms like the Bulkley Valley, or the Peace River Valley around Fort St. John. Unfortunately, these areas of higher risk often correspond with prime moose winter habitat and water bird nesting habitat.

Interior Rainforest (ICH, wet SBS)
The Interior wet-belt has wetland ecosystems that blend interior and coastal qualities. Though the cool, wet climatic conditions are conducive to wetland formation, the primarily mountainous terrain results in smaller numbers than expected. Large peatlands or swamps complexes are found in a few drainages such as the Seymour, Adams, Torpy, and Hominka rivers. Unfortunately, in most other drainages, the wetlands are few and share valley bottoms with road networks and agricultural development. Although we cannot suggest specific wetland communities at risk in this region, we can suggest that their natural scarcity combined with increasing development pressures may place most of these habitats and their dependents at risk.

Taiga Plains (BWBSmw)
The northeast corner of the province is subarctic and underlain by poorly drained, glaciolacustrine deposits. Consequently, this region has some of the highest concentrations of wetlands (largely peatland) in the province. Bogs predominate though fens and swamps occur along the sluggish streams that drain the region. Oil and gas exploration and development impact on wetlands but affect a relatively small percentage of the total wetland area. Forest harvesting of the most productive habitats, riparian white spruce (*Picea glauca*) and balsam poplar (*Populus balsamifera*) alluvial forest sites, may be of greatest concern as these provide very important wildlife habitat.

Interior Mountains (ESSF)
The mountain region has relatively few wetlands because of steep terrain. However, wetlands do occur in depressions and occasionally at slope breaks where springs or snow seepage keeps sites permanently saturated. Communities in these cold continental climates are often unique and do not occur at lower elevations. Risks are not high in this region though increased forestry activities at higher elevations may impact some areas.

Windward Coast Mountains (MH)
Like the hyperoceanic lowlands, the windward Coast Mountains are mineral-poor granite and experience extreme precipitation. These higher elevation wetlands have similar species as their lower elevation counter parts but form distinct assemblages. These sites are remote and exhibit little or no resource development potential so risks to wetlands are rare.

Northern Plateaus (SWB)
Some of the northern, subalpine plateaus of the northwest, such as the Kawdy Plateau, have a high concentration of wetlands. These sites provide important waterfowl nesting habitat for species such as lesser scaup (*Aythya affinis*) and northern pintail (*Anas acuta*) (Hawkings and Majiski 1991). At this time, mostly because of their remote location, risks to these wetlands are few.

What Is a Wetland?

Clearly defining the ecosystems is one of the tasks of the wetland classification program. The program has adopted the wetland definition used in the *Riparian Management Area Guidebook* (B.C. Ministry of Forests and BC Environment 1995). The guidebook recognizes that wetlands occur where soils are water-saturated for a sufficient length of time such that excess water and resulting low soil-oxygen

levels are principal determinants of vegetation and soil development. Specifically, "wetlands" must have both:

- plant communities characterized by species that normally grow in soils water-saturated for a major portion of the growing season ("hydrophytes"); and
- soils with surface peat ("O") horizons or gleyed mineral horizons (Bg or Cg) within 30 cm of the soil surface.

The term "riparian" is used in WREC as a general description of adjacency and not as a specific type of ecosystem. The term only means that the ecosystem, regardless of it composition, occurs next to a water body or wetland. Therefore, any type of ecosystem may be "riparian." In WREC, ecosystems that are ecologically distinct because of flooding, erosion/sedimentation, or subirrigation from an adjacent water body are called "flood ecosystems."

"Flood ecosystems" are sites flooded for short duration during the growing season where soils are freely drained and anoxic conditions (if they occur) are quickly relieved after subsidence of floodwaters. Plant species tolerant of brief flooding events but not prolonged soil saturation are typical. Flood ecosystems occur specifically where:

- water bodies periodically flood their banks depositing or eroding fluvial or lacustrine materials; and
- water tables are within the rooting zone during part of the growing season, but not for sufficient duration to cause gleying within 30 cm of the soil surface.

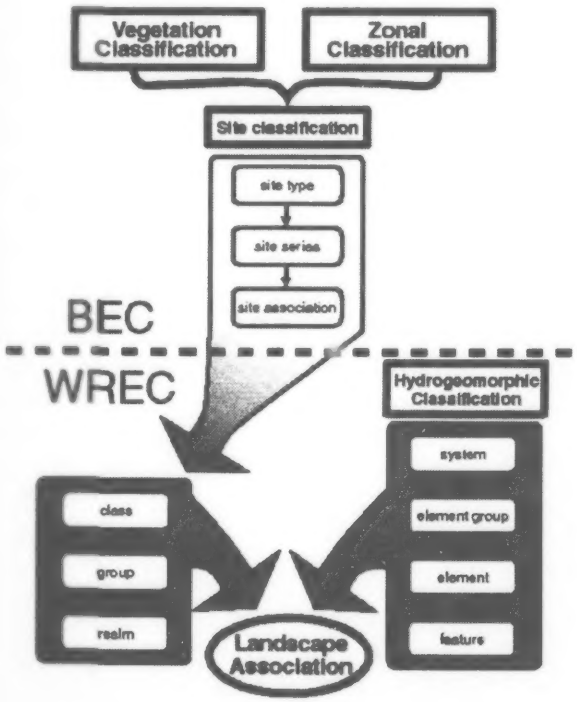

Figure 2. The Wetland and Riparian Ecosystem Classification (WREC) framework.

Classification Approach

The approach of WREC proposes a framework that places wetland ecosystems and riparian areas into a system that reflects the importance of landscape context, hydrological processes, and biological structure and composition. To this end, WREC incorporates 3 components arranged in a hierarchical structure (Fig. 2):

- an ecosystem component that describes site potential for biological communities;
- a hydrogeomorphic component that describes broad hydrological processes and concurrent geomorphic forms; and
- an integration component that unites the ecosystem and hydrogeomorphic components into associations at the landscape scale.

The basis of WREC is the biogeoclimatic ecosystem classification (BEC) of Pojar et al. (1987). The working framework uses BEC's zonal and site classification as its template. However, wetland and riparian ecosystems differ from the upland ecosystems described by BEC in 3 significant ways:

- hydrology and hydrodynamics are major site factors;
- sites are intimately connected to "upstream" landscape processes through hydrology; and
- sites are rarely homogenous except at fine spatial scales (<1:5,000) due to dynamic hydrological processes or internal environmental mosaics.

To accommodate these differences, several modifications have been made to BEC for its application to wetland and riparian ecosystems. Units broader than the site association are added to bring the site concept in line with the widely used Wetland Class unit of the National Wetlands Working Group (NWWG 1997). Hydrological criteria are also proposed as basic defining environmental site factors and as a component of a broader landscape unit classification (MacKenzie and Banner in prep.).

Ecosystem Classification

In WREC, specific ecosystems are successively grouped into more broadly affiliated units based on biologically relevant site characteristics (e.g., similar site associations placed in a single class). Functional units are defined by site potential and characterized by certain guilds of biota. Units emphasize similarities in basic underlying processes and provide a means of relating ecosystems at multiple scales. Each of these levels can have different applications (Table 1). For example, the site association defines site potential using climax communities. These units are specific and require a certain level of botanical knowledge to be applied successfully. For some applications such as mapping at a broad scale, evaluating initial habitat, establishing comparative research trials,

Table 1. Interpretive characteristics for each ecosystem of the Wetland and Riparian Ecosystem Classification (WREC).

Ecosystem Unit	Possible uses of the level	Example of the Unit	Characteristics of the example
Realm	Identification of appropriate classification structure to use, important environmental factors and broad biotic groups.	Terrestrial	Site is likely dominated by upland vascular plants typical in the climatic region and is on relatively well-drained soils.
Group	Identification of ecosystems with a common dominant ecological factor that will influence management and research. Blocking variable in research design and management interpretations	Flood	Site is riparian and is at least periodically flooded. Community dominated by non-wetland plants tolerant of flood events. Possible high wildlife capability and productivity sites.
Class	Broad management interpretations based on known characteristics. Prioritizing sites for habitat protection. Extension and communication of results to untrained users.	Low Bench	Shrub community with a sparse understory occurring directly adjacent to flowing water. The site is not a wetland but experiences long periods of flooding with pronounced erosion and deposition. Commercial tree growth not possible. Prolonged spring flood render sites unsuitable for ground-nesters and burrowers.
Site Association	Identification of rare or sensitive ecosystems. Specific wildlife or fisheries habitat capability interpretation. Specific management interpretations based on known controlling site factors. Distribution.	Drummond's Willow – Bluejoint	Tall shrub community of Drummond's willow, and erosion resistant graminoids and annuals. Site likely on sandy/silty levees beside slow moving, interior streams.

and communicating results to users with little botanical knowledge, broader ecosystem groupings may be preferred.

The classification's higher ecosystem levels (class, group, and realm) provide a consistent means of grouping site associations with similar underlying features, emphasizing similarities between ecosystems. Table 1 outlines information that can be derived from each level of this ecosystem hierarchy.

In BEC, site series are related to site conditions within a climate area (subzone or variant) using an edatopic grid. The grid illustrates the relationship of the site series to soil moisture and nutrient regime. However, the influence of hydrology in wetland ecosystems is not well represented in this model, making placement of these ecosystems problematic. For this reason, WREC introduces a modified edatopic grid, which includes hydrological variables in the subhydric and hydric moisture regime position of BEC's edatopic grid (Fig. 3). The hydrological variables considered are pH (as an indicator of base cation availability) and "hydrodynamic" index (degree and influence of water movements on the site).

Hydrogeomorphic Classification

The hydrogeomorphic component is a classification of hydrological and geomorphic features, which are fundamental features controlling ecosystem structure and composition in wetland and flood ecosystems. Historically, hydrogeomorphic classification has not been used in British Columbia but is the central concept of wetland identification and management in the United States (Cowardin et al. 1979, Brinson 1993, Bedford 1996). Most of these units rely on knowledge of landscape properties rather than specific hydrological variables and are therefore easily assessed (Bedford 1996). For purposes such as monitoring water quality or fisheries value, the hydrogeomorphic classification may be more appropriate than the ecosystem classification (Brinson 1993).

The WREC outlines a hierarchy then defines hydrological systems (as the broadest unit) and their associated geomorphological forms (Fig. 2). The system describes broad patterns of hydrology, water source, and topography. Wetland and related ecosystems occur within 4 different hydrological systems (Table 2).

The hydrogeomorphic component of WREC puts sites within a larger hydrological context. For example, the Drummond's willow (*Salix drummondiana*)–Bluejoint (*Calamagrostis canadensis*) site association described in Table 1 frequently occurs within the hydrological template described in Table 3. Depending on the hydrogeomorphic unit considered, one to many ecosystems may occur within it. The element level of the hydrogeomorphic classification describes an entire wetland or riparian landscape unit and would typically include several distinct site associations.

Ecosystems that are repeatedly associated within a hydrological template are termed a landscape association in WREC (MacKenzie and Banner in prep.).

Other hydrological features of importance in wetlands such as period of inundation, duration of soil saturation, pH, hydrodynamic index, and salinity are not directly part of the hydrogeomorphic classification but are used to define site associations.

CLASSIFICATION AS A TOOL FOR MANAGING HABITATS AT RISK

Noss (1994) observes a lack of understanding regarding the relationship between human activities and loss of diversity. Over the last 2 decades, scientists have begun to identify the critical role wetlands play for wildlife populations such as ducks and fish. However, the tools to study and understand

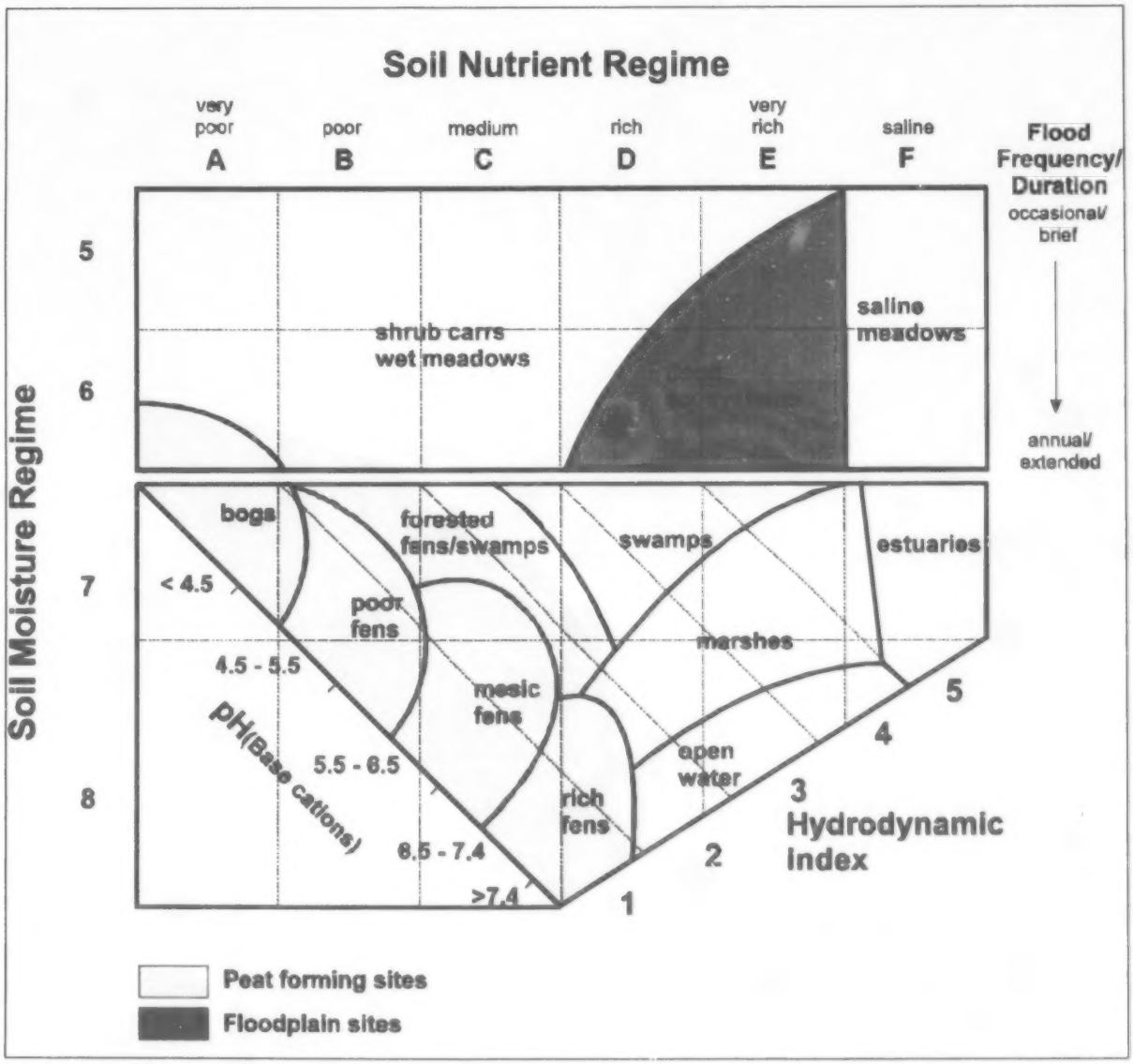

Figure 3. Relationship of wetland classes on a modified edatopic grid.

Table 2. Characteristics of 3 hydrogeomorphic systems of the Wetland and Riparian Ecosystem Classification (WREC) framework.

System	Topographic position	System Hydrology	Currently described Element Groups
Estuary	Confluence of freshwater inflow into marine environment	Subject to diurnal or periodic flooding and brackish water	Enclosed, Constrained, Extruded
Lacustrine	Adjacent to large, deep lakes	Subject to flooding by wave action; fed by circulating lake waters	Anthropogenic, Littoral, Deepwater, Peatland
Fluvial	Adjacent to flowing water course	Subject to annual stream flooding and erosion/ deposition forces; stream-fed	Alluvial, Transport, Headwater
Palustrine	Depressions or receiving sites; small shallow lakes, basins, and seepage slopes	Low energy flooding or groundwater-fed	Closed Basin, Overflow Basin, Linked Basin, Terminal Basin, Overflow Hollow, Linked Hollow, Toe Slope, Delta slope, Blanket Slope

wetlands have been scarce. One helpful tool is a comprehensive classification of wetland and riparian ecosystems. We identify several areas where WREC could be used for managing wetlands as habitats at risk.

- **Identification of natural plant communities.** The site associations of WREC are natural plant communities. Disturbed wetland sites were not used in the classification. The site association can therefore be used as a benchmark for restoration efforts or for assessing and recognizing impacts from land use practices.
- **Reference sites and hydrogeomorphic profiles.** A hydrogeomorphic profile for "fully functioning" environments provides the basic template to which potentially impacted wetlands can be compared (Brinson 1993, Bedford 1996). Brinson and Rheinhardt (1996) have promoted the use of reference wetlands for monitoring and researching potential impacts and mitigation. In addition, monitoring vegetation changes over time and comparing them with known site associations can indicate the success of restoration efforts
- **Identification of habitat value.** Known wildlife preferences for wetland plant species, structure, or hydrological form could be used to rank habitat values of different wetland types.
- **Identification of appropriate management techniques.** An understanding of sensitivities, functions, and values of different wetland types can identify appropriate and inappropriate management strategies. Risk and value of different wetland types could be ranked to help guide planning.

Table 3. Some characteristics and interpretations made for different levels of the hydrogeomorphic classification of the Wetland and Riparian Ecosystem Classification (WREC).

Hydrogeomorphic Unit	Example	Characteristics of example	Possible uses and interpretations
System	Fluvial	Site is associated with a river or stream. Water flow is unidirectional and likely seasonally or periodically variable (flood-pulse)	Special riparian management practices Possible high fisheries and wildlife habitat values.
Element group	Alluvial	Site is of low gradient and is a sediment deposition area. Likely with a well developed floodplain.	Flooding of riparian zone likely. Special development guidelines and wildlife values
Element	Tortuous Meander	Site very low gradient and has pronounced meanders and low width to depth ratio. May have oxbows, back channels, and back levee depressions. Very well developed floodplain.	System stable under normal conditions but sensitive to disturbance. Vegetation control of channel high and erosion potential high.
Feature	Levee	Raised ridge of fluvium usually directly adjacent to slow moving sediment laden stream coarse. Often downstream of higher gradient reaches	Specific environmental conditions for vegetation communities.

- **Inventory.** The classification provides a comprehensive system of recognized site associations and hydrological templates known to occur in the province. An inventory using wetland classification units can provide more accurate information regarding distribution, rarity, and landscape-scale impacts of land management practices
- **Gap analysis and coarse-filter protection for species at risk.** Certain wildlife or plant species of concern are likely associated with specific wetland types. Identification of this relationship could allow selective protection of those wetlands providing the best habitat. This approach has been demonstrated as an effective coarse-filter approach to protecting species at risk (e.g., Panzer and Schwartz 1998)
- **Direct further research.** Ecosystem classification is valuable in ecological research. Experimental design incorporating ecosystem units allows research to address the influence of different ecological attributes and to transfer knowledge from one location to others with similar attributes. Adaptive management approaches also require classification to direct management practice to appropriate site types.

CONCLUSIONS

Classifications, like any sound management tool, must be able to address varying levels of detail, from global and continental scales to regional and local scales. In British Columbia, we have been developing a hierarchical classification framework to create a comprehensive provincial wetland and riparian ecosystem classification. At present, British Columbia's wetland management policies are based on ecologically simple units, largely due to a lack of sound baseline ecological knowledge of these sites. The classification aims to assist researchers, managers, students, and resource workers to understand these ecosystems. Where development and resource extraction threaten to disturb wetlands, the classification can be used to help improve management and planning strategies.

Several major directions now await the Wetland And Riparian Ecosystem Classification program:

1. Completing landscape associations that recognize repeating complexes of site associations in relation to hydrogeomorphic form.
2. Applying WREC to management and wildlife habitat issues.
3. Using WREC with existing forest cover, TRIM database, and TEM mapping (where available) to create a preliminary provincial wetland inventory.

ACKNOWLEDGEMENTS

We thank Forest Renewal British Columbia for its generous funding of this project and the support of anonymous Science Council project reviewers. A. Banner, D. Meidinger, J. Pojar, R. Coupe, D. Lloyd, and many other ecologists have contributed invaluable expertise and historical context to this project. M. Kranabetter provided valuable and insightful comments on this paper.

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Biomonitoring with the Reference Condition Approach for the Detection of Aquatic Ecosystems at Risk

Pamela F. Reece
Department of Forest Sciences
2424 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada
pamreece@interchange.ubc.ca

John S. Richardson
British Columbia Ministry of Environment, Lands and Parks
and Department of Forest Sciences
2424 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

One of the most effective techniques for biomonitoring and detecting habitats at risk in aquatic ecosystems is the reference condition approach (RCA). The RCA uses benthic community structure as a measure of a system's condition. With the RCA, the benthic community of a potentially stressed ecosystem is compared with that of unstressed reference sites that have similar environmental conditions. Benthos have several advantages over other indicators: there are many species; they are relatively sedentary; and they represent a diverse range of trophic positions and tolerance to disturbance. The RCA will soon be used for monitoring in the Fraser River basin (FRB) based on data from 219 reference sites in the FRB. The reference collections were made in early autumn and 1 question is the extent to which predictions of expected communities at test sites are dependent on the season when samples are collected. Results showed that the RCA was reasonably robust, but changes to the benthic community between seasons were large enough for us to recommend collecting test samples in the autumn.

Key words: aquatic, benthic, biomonitoring, Fraser River, invertebrate, reference condition approach, seasonal variation, streams.

Anthropogenic impacts that can stress aquatic ecosystems and place them at risk come from a multitude of point and non-point sources. Because of the multitude of possible impacts and response levels, there is a need for effective monitoring techniques and remediation. Biomonitoring uses biological responses to identify and monitor changes in the environment. In aquatic ecosystems, biomonitoring can be used to monitor changes in water quality, changes in the stream habitat, or even changes to surrounding watersheds. Use of biological responses to identify changes in an environment gives biological relevance to those changes.

This paper provides an introduction to what biomonitoring is and why benthic invertebrates are favoured organisms for biomonitoring, and a brief overview of some biomonitoring techniques. The reference condition approach (RCA) as a biomonitoring technique is discussed, including how the predictive model is built, the way the reference condition approach is applied in the Fraser River basin (FRB), and limitations due to seasonal variation of benthic invertebrate communities.

BIOMONITORING

Invertebrates as Indicators

Biomonitoring requires the selection of 1 organism or a suite of organisms whose response to changes in the environment will be monitored. Although plankton, fish, and invertebrates can all be used for biomonitoring in aquatic ecosystems, benthic invertebrates are the most frequently used. There are several reasons for this bias. First, they tend to move very little and are therefore representative of the area in which they are collected. Second, invertebrate life cycles are relatively short compared to fish and will therefore more quickly reflect changes in the environment through changes to population and community structure. Third, invertebrates live and feed in, on, and around sediments where toxins tend to accumulate. Because of this, the benthos themselves accumulate toxins and pass them up the food chain (Reynoldson 1987, Reice and Wohlenberg 1993). Fourth, benthic organisms vary in sensitivity to stressors and may respond to pollutants in the water column as well as those in the sediments (Schindler 1987). Finally, invertebrates are important components of the ecosystem. They are the primary food source of many fish, and play a critical role in the breaking down of organic matter and nutrient cycling.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Biomonitoring Techniques

There are several different biomonitoring techniques employed in aquatic ecosystems. The selection of an appropriate technique depends on the issues being addressed and available resources. Potential biomonitoring techniques include use of indicator species, biotic indices, rapid bioassessment, and predictive models. Use of indicator species for biomonitoring involves monitoring of invertebrate abundance, life cycles, reproductive success, and/or morphological deformities. Biotic indices use presence/absence and abundance of taxa with different levels of stress tolerance to monitor changes in the environment. Biomonitoring using rapid bioassessment techniques can include measures of richness, enumeration, community diversity measures, biotic indices, or combinations of these measures. When biotic indices are used in rapid bioassessment, only a few habitats may be sampled, few samples are collected, the samples may be small, and the invertebrates are identified to higher taxonomic levels. Predictive models for biomonitoring use multivariate techniques, and information on benthic macroinvertebrate communities and associated environmental variables from a large number of reference sites to make predictions as to the expected benthic community at test sites. Predictive models are proving to be effective for biomonitoring. The reference condition approach is an example of a predictive model.

THE REFERENCE CONDITION APPROACH

One of the most effective techniques for biomonitoring and detecting habitats at risk in aquatic ecosystems (or their watersheds) is the reference condition approach. The RCA involves creating a predictive model from benthic invertebrate and associated environmental data collected from a large number of reference sites. Reference samples from a large number of habitats allows a wide variety of habitats and invertebrate communities to be included in the model. The model can then be used to predict the expected benthic community at a test site. If the test-site community differs from the one predicted, the conclusion can be drawn that the site is impacted.

The reference condition approach is being used for biomonitoring and detecting habitats at risk in the Fraser River basin. The Fraser River basin predictive model was created using benthic invertebrate data and associated environmental conditions collected from 219 reference sites from throughout the FRB (Reynoldson et al. in prog.). Sites were selected as reference sites if they had no or minimal impacts (Rosenberg et al. in prog.)

In general, creation of the predictive model involves identifying reference groups with similar species composition, identifying which environmental variables can best discriminate between the reference groups, and then using the environmental variables to predict a test site to a reference group. Cluster analysis of the reference-site invertebrate data is used to form reference groups (Fig. 1). As a result, each reference group contains sites with similar species composition, and sites are allocated to reference groups objectively. Based on the reference groups, discriminant function analysis (DFA) can then be used to identify environmental variables that best discriminate between the reference groups, and can therefore be used as predictor variables (Fig. 1).

Once the environmental predictor variables are selected, DFA can then be used to predict a test site to a reference group. A test site is defined as a site which is meant for comparison with the reference condition. DFA uses the environmental conditions from the test site, to predict the site to a reference group with similar environmental conditions. The test site is then expected to have a benthic invertebrate community composition similar to the reference sites included in the reference group. If the actual test-site community differs from the one predicted, then the conclusion can be drawn that the system is stressed or impacted.

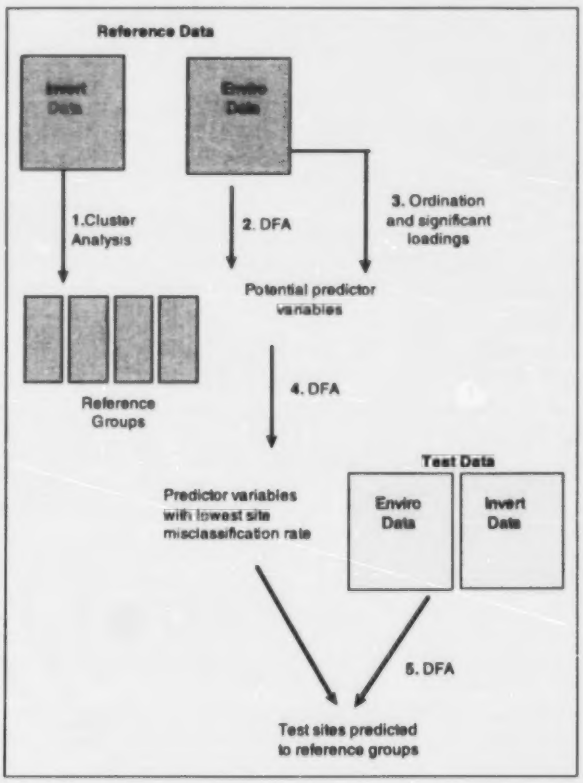

Figure 1. An overview of the steps required to make a predictive model and to predict test sites to reference groups. (DFA = discriminant function analysis.)

Similarity of a test-site invertebrate community to the reference condition is determined using probability ellipses. Invertebrate data for each reference group are ordinated. Probability ellipses of 90, 99, and 99.9% are then calculated based on ordination scores of the reference sites included in a reference group. Use of probability ellipses allows invertebrate community responses to be viewed along a gradient of response levels—from reference to potential impact-response levels. After the probability ellipses are calculated, the ordination score of the test site is plotted in the reference group ordination space. If the test site falls within the 90% ellipse, the site is considered equivalent to reference. If the test site falls between the 90 and 99% ellipses the site is considered to be potentially different from the reference condition. A test site outside of the 99% ellipse is different from the reference condition, and one outside of the 99.9% ellipse is very different from the reference condition.

Limitations of the RCA Due to Seasonal Change of Invertebrate Communities

Accurate predictions as to the expected benthic community are limited to the range of environmental conditions and benthic data included in the reference database. The predictive model constructed for the FRB contains a large amount of spatial variation, but at this point in time it does not contain seasonal variation. The reference samples were collected only once a year—in the autumn—but over 3 years (1994, 1995, and 1996). There was the potential that sampling only once in a year may not provide enough information to accurately apply the model to test samples collected at other times of the year. This is because the benthic community may change seasonally as invertebrates move though their life cycles.

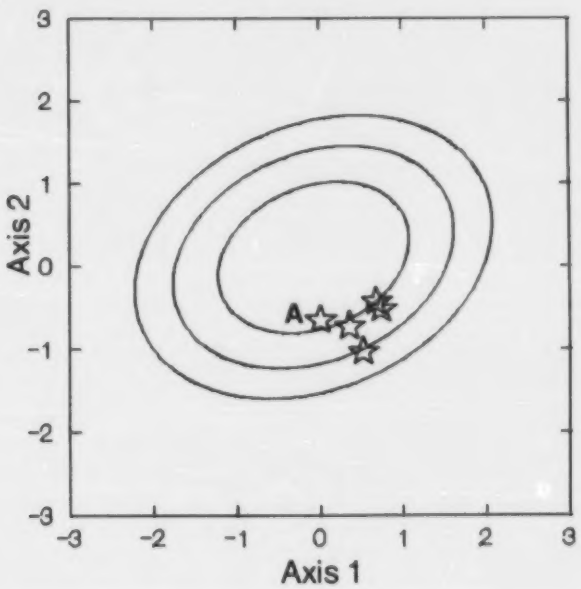

Figure 2. Location of Spring Creek seasonal samples in the ordination space of the reference group into which the site was classified based on the autumn (A) sampling. The invertebrates were identified to the genus level. The 90, 99, and 99.9% probability ellipses are based on the ordination scores of the reference data.

We looked at what implications seasonal variation in the benthic community had on using the RCA in the FRB by collecting seasonal samples from 6 reference sites. The samples were collected over a period of 1 year: in the spring, summer, autumn, and winter, and then a second spring. The seasonal invertebrate data for each site were ordinated and the points plotted in the respective reference group ordination spaces. Figure 2 illustrates the results for Spring Creek, and Table 1 summarizes the results for all 6 sites, (invertebrates were identified to the lowest possible taxonomic level; Reece et al. in prog.).

Our study and others have shown that stream benthic invertebrate community composition changes seasonally (Hynes 1970, Furse et al 1984). Seasonal changes of the invertebrate communities were enough to place some of the sampling dates outside of the reference ellipses. The magnitude and direction of the seasonal change in ordination space was not consistent between sites or seasons and creates uncertainty. The uncertainty comes from the inability to separate typical seasonal changes from changes caused by stress on the system. Thus seasonal changes of the invertebrate community affect how the Fraser River predictive model should be used.

When using the Fraser River predictive model, test samples should be collected in the same season as when the reference samples were collected (autumn). This should prevent misclassification of the state of a test site simply due to seasonal variation. In future, reference samples could be collected over multiple seasons and added to the reference database. This would allow seasonal variation to be incorporated into the model.

CONCLUSIONS

In conclusion, use of biomonitoring for the measurement of changes to an ecosystem gives biological relevance to those changes. In aquatic ecosystems, invertebrates have been the most commonly used organisms for biomonitoring and there are several good reasons for this preference. Once an initial reference database is established, the reference condition approach is a quick and accurate technique for detecting changes in benthic invertebrate communities. In the Fraser River basin, test samples should be collected in the autumn—when reference samples were collected. This will

Table 1. Summary of the reference groups into which the seasonal test sites were classified based on the autumn sampling, and where the seasons fall in the ordination space. Invertebrates were identified to the genus level.

Study site	Season	Reference group	Location[a]
Glimpse	spring '95	5	>99.9
	summer	5	>99.9
	autumn	5	in
	winter	5	>90
	spring '96	5	>90
Beak	spring '95	5	in
	summer	5	in
	autumn	5	in
	spring '96	5	in
Spring	spring '95	6	>90
	summer	6	>90
	autumn	6	in
	winter	6	>99
	spring '96	6	>90
Mayfly	spring '95	2	in
	summer	2	>90
	autumn	2	>90
	winter	2	>90
	spring '96	2	>90
North Alouette	spring '95	2	>90
	summer	2	in
	autumn	2	in
	winter	2	in
	spring '96	2	in
Thompson	spring '95	2	>99
	summer	2	>90
	autumn	2	>90
	winter	2	in
	spring '96	2	>99

[a] in = within the 90% probability ellipse; >90 = between the 90 and 99% ellipses; >99 = between the 99 and 99.9% ellipses; >99.9 = outside the 99.9% ellipse.

minimize misclassification of test sites until seasonal reference site data can be added to the reference database.

ACKNOWLEDGEMENTS

We thank T. Tuominen, and Environment Canada's Fraser River Action Plan for support of the project. J. Detman, J. Kaman, R. Letchford, C. Logan, and J. Melody helped with sampling, sample processing, and invertebrate identification of the seasonal samples. A. Borkent, J. Cumming, D. Currie, and B. Kondratieff generously donated their time and effort to verify invertebrate identifications.

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Monitoring Riparian Leave Strips with Multi-Temporal RADARSAT C-Band Satellite Data

Peter A. Murtha
Forest Resources Management, University of British Columbia
2045-2424 Main Mall, Vancouver, BC, V6T 1Z4, Canada
murtha@interchg.ubc.ca

ABSTRACT

The British Columbia Forest Practices Code (FPC) requires protection of fish-bearing river and stream reaches through riparian reserve zone strips. This paper reports on the application of Fine 2 beam mode RADARSAT multi-temporal data to monitoring 65 riparian strips. Eleven Fine 2 beam mode RADARSAT images were acquired between 4 December 1996 and 24 March 1999; these were examined individually and as multi-temporal colour composites (MTCs) for strip-change detection. The 65 riparian strips have a total length of 24.4 km. Eighteen strips comprising 6.3 km (25.8%) are intact. The remaining 47 strips have holes of varying length. The measurement of holes shows that by November 1998, 5.56 km (30.5%) of the 18.2 km of the post-FPC strip length has been decimated by windstorms, and that overall, 7.785 km (31.85%) of the total strip length is decimated. Most decimated strips are found on till-based landforms.

Key words: clearcuts, Forest Practices Code, RADARSAT, riparian leave strips, Vancouver Island, windthrow.

The British Columbia Forest Practices Code (FPC) provides guidelines for forest operations and silvicultural prescriptions, and requires protection of fish-bearing river and stream reaches through a system of riparian reserve and management zones (collectively termed riparian leave strips; B.C. MOF and B.C. MELP 1993). Fish-bearing streams <1.5 m wide have a management zone 30 m wide, while non-fish-bearing streams <3 m wide have a management zone of 20 m. The riparian leave strips are found in or alongside new clearcuts. On northern Vancouver Island these strips are subject to severe wind and rain storms. Many of the strips are quickly decimated through windthrow (Fig. 1). Strips range in width from 20 to 70 m, and may be as short as 60 m

Figure 1. Riparian strip 16 as it looked on 20 August 1997. This strip was created in August 1995. Aerial photographs (Fig. 2) taken at the same time showed 55 codominant tree crowns in the strip. This strip was decimated by windstorms during the autumn–winter of 1995–96. The strip was 300 m in length. (Photo by P. Murtha.)

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

and as long as 700 m. RADARSAT Fine 2 beam mode data can image these strips, and their detectability depends on strip condition (Murtha and Mitchell 1998).

RADARSAT is a Canadian satellite that was launched in November 1995. It has a C-band (wavelength 5.3 cm) radar sensor, and multi-look and beam modes. Orbiting at an altitude of about 800 km, in a north–south polar orbit, it has a 24-day repeat cycle. Since it is radar, it is possible to obtain data from the same orbital position once every 24 days, regardless of darkness or weather conditions. More details about RADARSAT and radar can be found in Lillesand and Kiefer (1994), and on the Web. (For the satellite description see RADARSAT International at http://www.rsi.ca/home.htm.)

Murtha and Mitchell (1998) reported that all of the heavily damaged and decimated strips are not imaged (Figs. 2 and 3), and that all of the healthy to medium-damaged strips are easy or moderately easy to locate on multi-temporal colour composites (MTCs). The visibility of the strips is also dependent upon contrast with their background (Murtha 1998). On northern Vancouver Island all post-FPC clearcuts are imaged bright during the rainy winter–spring months. When the weather becomes drier in the summer, the clearcuts are imaged dark. Clearcuts on glacial till landforms, such as ground moraine, and till over bedrock remain brighter longer than do clearcuts on marine or lacustrine deposits (lake bed, lake bed terraces), glacial fluvial outwash deposits, or flood plains (Murtha 1998). The MTCs made the strips easier to interpret regardless of the season.

This paper reports on the RADARSAT monitoring of riparian leave strip decimation on northern Vancouver Island.

STUDY AREA

The study area is located on northern Vancouver Island, B.C., between the communities of Port Hardy and Port McNeill (Murtha and Pollock 1996). It is located in the Nahwitti Lowlands ecosection and in the Coastal Western Hemlock (CWH) biogeoclimatic zone (Klinka et al. 1991, Krajina 1965). The surficial geology has been mapped

Figure 2. August 1995 aerial photograph showing intact riparian strip 16 bisecting the cutblock. When viewed stereoscopically, 55 codominant tree crowns can be counted. (Aerial photography by Foto Flight Surveys, Calgary, AB.)

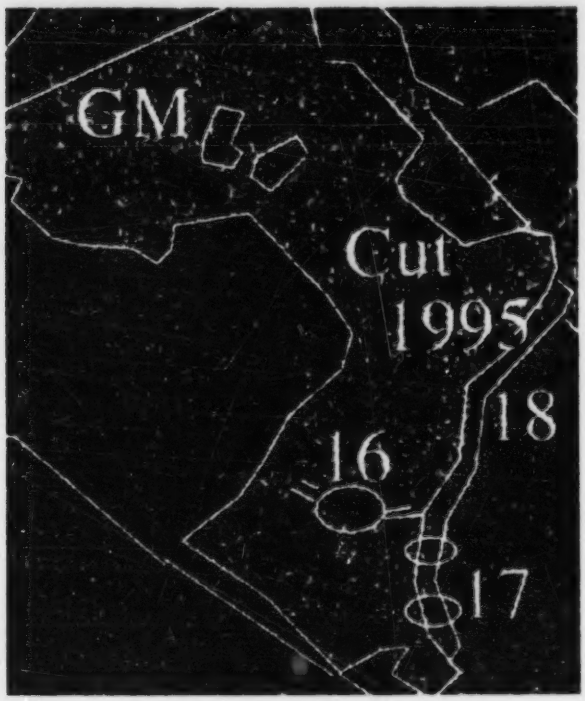

Figure 3. The first RADARSAT Fine 2 beam mode images showed no trace of riparian strip 16. Even though the image data were acquired at night during rainstorms, riparian strips 17 and 18 on the right side of the cutblock are clearly evident. Decimated riparian strips do not show on the RADARSAT images. (RADARSAT images copyright Canadian Space Agency, 1996, 1997, and 1998.)

according to textures, genetic materials, surface forms, and modifying processes (Lewis 1985). The dominant tree species are Sitka spruce (*Picea sitchensis*), western hemlock (*Tsuga heterophylla*), western redcedar (*Thuja plicata*), and amabilis fir (*Abies amabilis*; Klinka et al. 1991). The study area borders the Port Hardy airport, where Environment Canada maintains a weather station.

METHODS

Eleven Fine 2 (F2) beam mode RADARSAT images were acquired between 4 December 1996 and 24 March 1999 (Table 1). All F2 RADARSAT data were acquired at 0221 hr UTC. F2 beam mode images are 50 km on the side, and have a nominal of 8-m pixel resolution and an incidence angle of about 40°. Thirteen 512 X 512 pixel subscenes were selected from each RADARSAT Fine 2 scene using ER Mapper 5.5 software. The subscene image pixel display resolution is 7.60 X 7.60 m. Each subscene was registered to NTS map sheet 92L/11, then each of the remaining 10 dates for each subscene were co-registered in order to create MTCs.

Fifty strips were located on 1:20,000 colour aerial photographs from 1995. Seventeen strips were pre-FPC (before 1994). The landform of the clearcuts was interpreted from the 1995 aerial photographs (Kesser 1976, Mollard 1978). The RADARSAT F2 images were examined individually and, after receipt of the first 3 images, as MTCs. The known riparian strips looked like sinuous, dark noodles in or alongside the bright cutblocks (Figs. 2 and 3). Fifteen new strips were found on the RADARSAT images. The length, width, and direction of the original 50 strips were measured on 1995 colour aerial photos. The RADARSAT F2 MTCs were used to measure the strips created after August 1995.

Holes in strips were measured on the aerial photographs and on the RADARSAT MTCs. As strips become more damaged, trees go down in groups, creating the holes. These holes are seen on the RADARSAT MTC images as strips that should have shadows, but don't, or the MTC-colour of the

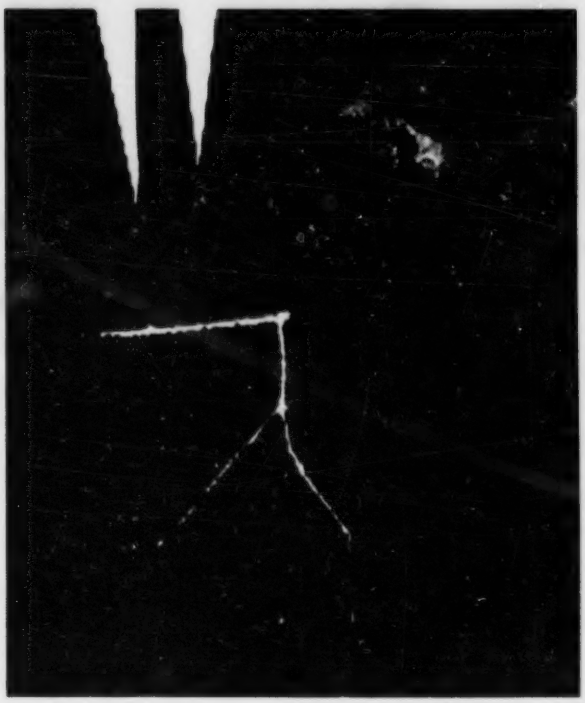

Figure 4. August 1995 aerial photograph showing riparian strip 31b. Pointers indicate holes, which measure a total of 4 mm in length. Stereoscopically, a windthrown tree is visible by the left-hand hole. At the original scale of 1:20,000, the lineal distance of holes equals 80 m. (Aerial Photography by Foto Flight Surveys, Calgary, AB.)

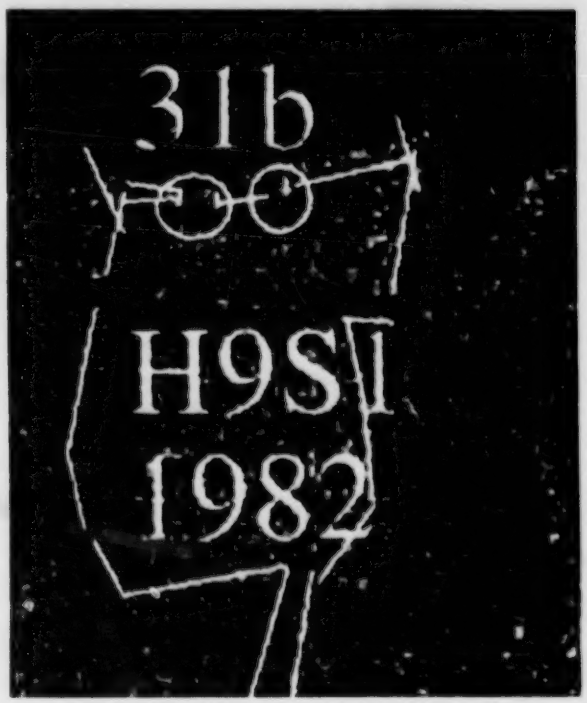

Figure 5. The holes in riparian strip 31b are circled in this RADARSAT image, acquired at night during rainstorms, from 800 km in space. Using ER Mapper image analysis software, the measured holes totalled 88 m. (H9S1–1982 indicates the cutblock was planted to 90% hemlock and 10% spruce.) (RADARSAT images copyright Canadian Space Agency, 1996, 1997, and 1998.)

Table 1. RADARSAT data acquired over northern Vancouver Island, B.C. from 4 December 1996 to 24 March 1999.

Date	RADARSAT data	Daylight	Weather
4 Dec 96	Fine 2	No	heavy rain, strong winds
11 Dec 96	Standard 4	No	heavy rain
14 Feb 97	Fine 2	No	heavy rain, windy
3 Apr 97	Fine 2	No	gale force winds, drizzle
8 Jul 97	Fine 2	Yes	drizzle
15 Jul 97	Standard 4	Yes	overcast, scattered drizzle
1 Aug 97	Fine 2	Yes	overcast, no rain for 6 hr
25 Aug 97	Fine 2	Yes	light rain
5 Nov 97	Fine 2	No	rain
12 Nov 97	Standard 4	No	rain
29 Nov 97	Fine 2	No	heavy rain
6 Dec 97	Standard 4	No	overcast, no rain
20 Aug 98	Fine 2	Yes	overcast, no rain
24 Nov 98[a]	Fine 2	No	rain
24 Mar 99[b]	Fine 2	No	frontal rains

[a] Image courtesy of British Columbia Ministry of Environment, Lands and Parks.

[b] These data were received too late to include strip analysis in this paper, however initial analysis shows continued decimation of the riparian strips.

cutblock bleeds through the strip when it shouldn't. The accuracy of the measurements was compared with the aerial photographs. For example, strip 31B had holes clearly evident on the aerial photographs (Fig. 4), with a cumulative length of 4 mm, or 80 m at a scale of 1:20,000. The same holes measured on the RADARSAT composites (Fig. 5) with ER Mapper software gave a length of 88 m. These tests indicate that measurements made on the RADARSAT MTC are within ±10% of measurements made on the aerial photographs. Holes in all strips were measured on images from 25 August 1997, 29 November 1997, 20 August 1998, and 24 November 1998.

RESULTS

Data on year of creation of the strips, landform of the cutblock, total strip length, and decimation (holes in strips) are presented in Tables 2 and 3. The 17 pre-FPC decimated strips are found primarily on ground moraine. Table 3 indicates that the newly decimated strips created since the implementation of the FPC are also found on ground moraine, or till with bedrock close to the surface. Better drained and having better site conditions, the till deposits support CWH climax species (western hemlock) because of their mesic site attributes. Rooting depth is more restricted on the till deposits. Thus when the adjacent forest is cut, the tall (30–50 m) hemlock are more predisposed to windthrow than are the cedar

Table 2. Pre-Forest Practices Code (before 1994) riparian strips classified according to year of cutting, landform, total length, and length of holes measured on the RADARSAT images.

Strip #	Year of cutting	Land- form[a]	Total length (m)	Length of holes (m)			
				25 Aug 97	29 Nov 97	20 Aug 98	24 Nov 98
04	1984	LB	160				20
10	1992	GM	240	240	240	240	240
11	1992	GM	100	100	100	100	100
12	1991	LB	580				40
14	1991	LB	800				224
19	1993	LB	800				
20	1993	LB	200				
22	1993	LB	200				
28	1985	LB	500				
29	1985	LB	320	320	320	320	320
30	1991	LB	400				
31B	1982	LB	400	88	88	141	199
32A1	1992	GM	300				
32A2	1992	GM	200	105	105	105	105
32B	1992	GM	180	180	180	180	180
32C	1992	GM	400	400	400	400	400
32D	1992	GM	400	400	400	400	400

[a] LB = lakebed; GM = ground moraine.

Table 3. Post-Forest Practices Code (1994 and later) riparian strips classified according to year of cutting, landform, total length, and length of holes measured on the RADARSAT images.

Strip #	Year of cutting	Land- form[a]	Total length (m)	Length of holes (m)			
				25 Aug 97	29 Nov 97	20 Aug 98	24 Nov 98
01	1995	GM	100	100	100	100	100
02	1995	GM	300	26	26	26	26
03	1995	GM	220				
05	1996	GM	400	20	20	20	20
06	1996	GM	200				
07	1996	GM	460	20	20	48	48
08	1996	GM	480				30
09	1994	LB	80				
13	1994	LB	320		30	30	57
16	1995	GM	300	300	300	300	300
17	1995	GM	400	94	94	134	176
18	1995	GM	440	95	95	153	202
21	1994	LB	100				
23	1994	LB	160				
24	1994	GM	300	300	300	300	300
25	1994	LB	1200	175	310	398	472
26A	1994	LB	200				
26B	1994	LB	200				
31A	1996	GM	440		60	80	113
33	1995	GM	280	280	280	280	280
34	1995	GM	140	140	140	140	140
35	1995	GM	400				
36	1995	T/R	800				230
36A1	1997	T/R	380		131	380	380
36A2	1997	T/R	320		60	320	320
37	1994	LB	900				146
38	1994	LB	220				
39	1995	T/R	500	82	82	82	80
40	1995	T/R	600			74	74
41	1995	T/R	500				
42	1995	T/R	300				
43	1995	T/R	260				
44	1995	T/R	300	90	90	90	90
45	1995	T/R	480			145	172
46A	1995	T/R	200	60	60	60	60
46B	1995	T/R	80	80	80	80	80
47	1995	T/R	300	167	167	167	167
48	1995	T/R	650	160	160	160	160
49A	1995	T/R	360	360	360	360	360
49B	1995	T/R	200	200	200	200	200
50	1997	GM	920		60	60	94
51	1997	LB	640				
52	1997	LB	560				50
53	1997	T/R	450				450
54	1997	T/R	200				45
55	1997	T/R	320				50
56	1997	T/R	380				44
7A	1996	GM	280				41

[a] LB = lakebed; GM = ground moraine; T/R = till/rock.

Table 4. Summary of data on length of holes measured in riparian strips from before and after implementation of the Forest Practices Code (FPC), as shown on RADARSAT F2 images.

Year of cutting	Total length of strips (m)	Length of holes (m) and % of total strip length			
		25 Aug 97	29 Nov 97	20 Aug 98	24 Nov 98
Before 1994 (pre-FPC strips)	6,180	1,833 / 29.66%	1,833 / 29.66%	1,886 / 30.5%	2,228 / 36.05%
1994 and later (post-FPC strips)	18,220	2,699 / 14.81%	3,225 / 17.7%	4,187 / 22.98%	5,557 / 30.5%
Total	24,400	4,582 / 18.78%	5,058 / 20.72%	6,073 / 24.88%	7,785 / 31.85%

on the wetter sites. These riparian strips are habitats at risk.

The 65 riparian strips, found in or beside 35 clearcuts, have a total length 24.4 km (Table 4). They also have an average length of 375 m, and an average width of 50 m. Of the 48 strips created since the implementation of the FPC in 1994, 30.5% of the total strip length has been decimated by windstorms. In the first images assessed, the total length of holes in all 65 strips combined was about 19% of total strip length. A year later (20 Aug 1998), the combined length of holes had increased by 6%, and at last measurement (24 Nov 1999) had increased by an additional 7%, for a total increase of 31.85%, or 7.78 km covering 38.92 ha (Table 4). (A RADARSAT F2 image acquired on 24 March 1999 shows further decimation of the strips I have been watching. The winter of 1998–99 has seen major windstorms. Undoubtedly strip decimation will have taken another jump when all the strip holes are remeasured.)

DISCUSSION

The riparian reserve zone and management zone leave strips on northern Vancouver Island are a by-product of the Forest Practices Code, and their numbers are increasing. The greater the numbers, the more difficult it is to monitor using traditional ground-based approaches. Clearly the riparian strips are being damaged by windstorms and are habitats at risk. The terrain is remote and somewhat difficult to access, which makes ground surveys slow and time-consuming, and thus very costly. Helicopter surveys cost about $1,000/hr, and leave no permanent record. Cloudy weather makes optical remote sensing questionable on a continually repetitive basis. Scheduling commercial aerial photography, and acquisition of aerial photographs on a continuing basis is currently not a option because of weather and the cost of aerial photography (>$50,000 for the Aug 1995 aerial photography). The Landsat thematic mapper (TM) can image the strips at a 25-m pixel resolution (which for some strips is inadequate resolution), but cloud cover precludes frequent imaging. The past 5 years of TM images illustrates the depth of the problem (see http://ceocat.ccrs.nrcan.gc.ca/cdql/landsat5/track-51/frame-25/); during the last 5.5 years about 18% of the images were cloud-free. The alternative is the Fine 2 beam mode of RADARSAT with an 8-m pixel resolution. The satellite turnaround time is remarkable. The image ordered from the 24 March 1999 satellite pass at 0221 hr UTC (= 23 Mar, 1921 hr PST, when a frontal storm system was passing over northern Vancouver Island) was acquired, processed, and delivered to the FIRMS lab in the Faculty of Forestry, University of British Columbia, then loaded into the computers prior to noon on 25 March: 38 hours between acquisition and application.

The critical time is during the winter storm period (Oct–Apr). This time is coincident with the best imaging period to obtain adequate contrast of strips against the bright background of the recent cuts. Riparian strips in the correct situation, like the Nahwitti Lowlands ecosection, can be monitored for the price of data acquisition and image processing. In the case of the 24 March 1999 image, it took less than half an hour to document further decimation of strips.

CONCLUSIONS

1. RADARSAT Fine 2 beam mode data can be used to assess decimation of riparian leave strips in clearcuts on northern Vancouver Island.
2. Thirty-five clearcuts had 65 riparian strips with a total length of 24.4 km.
3. 30.5% (5.6 km) of post-FPC strip length has been decimated.
4. The 7.78 km of strip decimated since 1990 covers 38.9 ha.
5. Post-FPC decimated strips are found on till-based landforms (till/rock, ground moraine).
6. Without the RADARSAT Fine 2 images, these data on damage to riparian leave strips would not have been readily available to this author or these proceedings. This report suggests a link between riparian strip decimation and the landform of the clearcut.

ACKNOWLEDGEMENTS

Cooperators in the study include Western Forest Products Ltd., RADARSAT International, the Canadian Space Agency, and the British Columbia Ministry of Environment, Lands and Parks. Special thanks are given to K. McGourlick (Western Forest Products Ltd., Port McNeill, B.C.). Image analysis and interpretation was done in the FIRMS Remote Sensing/GIS lab, Faculty of Forestry, University of British Columbia. Canadian Space Agency ADRO 384 funding is gratefully acknowledged.

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British Columbia's Freshwater Fish, Species, and Ecosystems Are More at Risk and Less Protected

Gordon R. Haas

British Columbia Ministry of Fisheries, Research Section, 2204 Main Mall, (and)
Centre for Biodiversity Research, Department of Zoology, and Native Fish Research Group
6270 University Boulevard, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada
Gordon.Haas@gems4.gov.bc.ca

ABSTRACT

Freshwater fish, other freshwater aquatic species, and freshwater ecosystems are not getting equal representation in conservation efforts in British Columbia compared to terrestrial species and ecosystems. They are much more often listed as at risk and Forest Dependent (FD) than any other taxonomic group and biogeographic or habitat type. This risk discrepancy is not addressed by protection under the British Columbia Forest Practices Code (FPC) or the British Columbia Protected Areas Strategy (PAS). Fish compose the highest proportion (one-third) of Red-listed (threatened or endangered) taxa, and 25% of these do not have proper taxonomic nomenclature. Sixty-seven percent of Red-listed fish have a global distribution of a single British Columbia water body, and a further 24% of Blue-listed (vulnerable or sensitive) and Red-listed fish are peripheral isolates. No commercial salmonid or other stocks are listed. Fish also have the highest percentage (82%) of taxa designated FD, which includes 35% of Red-listed fish. In contrast, less than 1% of FD fish taxa receive designated protection as Managed Identified Wildlife (MIW) under the FPC. Similarly, fish represent 25% of all vertebrates designated FD, but compose only 2% of the MIW listings for vertebrates. This protection presently amounts to a single fish species (bull trout), which is still likely inadequately covered by the standard MIW protective measure of a Wildlife Habitat Area (WHA), due to the inherent interconnectivity of watersheds. Without even considering commercial salmonids and fishes, there are at least 10 additional fish taxa strongly deserving forestry consideration. The PAS protects only (and often incidentally) 15% of Red-listed, 9% of Blue-listed, and 4% of FD fish taxa. About 14% of fish biodiversity "hotspots" are also protected. The PAS is based on conserving species through protection of the representative biogeoclimatic ecosections they live in. Ecosections (and WHAs) are a terrestrial concept and can have limited correspondence to the distribution of fish and aquatic biodiversity or their drainage basins. Fish and aquatic biodiversity are also uniquely composed of strong within-species variability resulting from both historical and ecological factors. Among fish, non-commercial or non-sport taxa are particularly poorly understood, with even basic taxonomic, ecological, and life-history information often not available. The management and conservation implications, which are substantial and immediate, are discussed in this paper. Equitable coordination of overall conservation is needed between aquatic and terrestrial biologists.

Key words: aquatic ecosystems, fish, Forest Practices Code, freshwater, Protected Areas Strategy, taxonomy.

The general public, and often even scientific, perception is that fish and aquatic species and ecosystems are receiving enough, or even too much, of the overall conservation effort and protection (McPhail 1993, Slaney et al. 1996, Haas 1998). This paper demonstrates that this is not the case, why it is not, and what might still be done about it. These points and the differences inherent to fish and aquatic ecosystem are analyzed through comparison to the other vertebrate taxonomic groups. The comparisons are based on:

1. Taxa Red-listed as threatened or endangered and Blue-listed as vulnerable or sensitive by the British Columbia Conservation Data Centre (CDC; Cannings and Ptolemy 1998, Smith 1998), and their global and provincial distributions in terms of limited occurrence.
2. Taxa designated as Forest Dependent (FD; Smith 1998), and other taxa worthy of consideration in terms of resource utilization impacts (e.g., Haas 1997*a*-*c*, 1998).
3. Protection under the British Columbia Protected Areas Strategy (PAS; Land Use Coordination Office 1999), Ecological Reserve System (B.C. MELP 1999*a*), and British Columbia Forest Practices Code (FPC; Forest Practices Code 1995, Haas 1998, B.C. MELP 1999*b*).
4. A preliminary examination from an aquatic perspective of

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

the utility of the biogeoclimatic terrestrial ecoprovinces and ecosections, which form the basis of the PAS (Land Use Coordination Office 1999). This is done in light of major drainage connections, their composite watershed groupings, aquatic biodiversity "hotspots," and the connectivity of aquatic systems.

5. A related examination of the unique patterns and magnitude of within-species variability in fish and aquatic species and ecosystems. In aquatic species, this variability is formed both by historical features largely corresponding to the last glacial period (10,000–15,000 years ago) and by ecological factors since then (McPhail and Carveth 1993*a*, Haas 1998).
6. Fundamental conservation and management knowledge about taxonomy, identification, ecology, and life-history (McPhail and Carveth 1993*a*, Haas 1998). This is also briefly examined from the viewpoint of commercial/sport versus non-commercial/non-sport fish taxa.
7. Recommendations for addressing these disparities, and for the conservation and management of fish and aquatic species and ecosystems, as well as general conservation and management (McPhail and Carveth 1993*a*, Haas 1998).

METHODS AND STUDY AREA

Overview analyses of the state of conservation, protection, and protected areas for freshwater fish, other freshwater aquatic species, and freshwater ecosystems throughout British Columbia were largely based on the latest data from Haas (1998) and that appreciatively made available in Cannings and Ptolemy (1998) and Smith (1998). The updated PAS and Ecological Reserve System information kindly came from D. Biffard (Parks Branch, B.C. Ministry of Environment, Lands and Parks). Additional information on these data are available at the web sites of the British Columbia Land Use Coordination Office (http://www.luco.gov.bc.ca), and the British Columbia Ministries of Environment, Lands and Parks (http://www.env.gov.bc.ca.), Fisheries (http://www.fisheries.gov.bc.ca), and Forests (http://www.for.gov.bc.ca.). All interpretation of this data and responsibility for it are nonetheless my own.

These data and analyses are specific to freshwater and anadromous fish in British Columbia. However, the interpretations and implications extend to freshwater fish, species, and ecosystems in at least the Pacific regions of western North America (e.g., Waples 1995, Allendorf et al. 1997), particularly in jurisdictions where parallels exist to the factors discussed here for British Columbia. These factors include poor or incomplete taxonomy and inventory, protected areas based on terrestrial ecosection or biogeoclimatic zone, and strong commercial- or sport-fish species biases and databases.

RESULTS

Risks

Red-listed Species

Of the vertebrate taxonomic groups designated and tracked by the Conservation Data Centre, fish compose the highest proportion of Red-listed taxa. Approximately one-third (32%) of freshwater fish taxa in British Columbia are Red-listed (Fig. 1). This ratio is about 10% higher than for reptiles (23%) or amphibians (22%), which constitute the taxonomic groups with the next highest proportions of Red-listed taxa. The combined total for all Red-listed non-fish vertebrates is only 9%. Fish also have proportionately more Red-listed taxa than non-vertebrate organisms such as plants. The term "taxa" is used as a species surrogate throughout this paper due to incomplete taxonomies and to remain consistent with its use in these listings by the CDC (Cannings and Ptolemy 1998).

Forest Dependency

Forest Dependent taxa are defined as those with a high to medium dependence on forests (full definitions and listings in Smith 1998). Fish compose the highest percentage (82%) of taxa that are officially designated as FD (Fig. 2; Smith 1998), and, of the taxonomic groups designated by the CDC, have the highest proportion of Red-listed taxa that are FD. Thirty-five percent of the Red-listed freshwater fish in British Columbia are FD. This ratio is 20% higher than for amphibians, which are the taxonomic group with the next highest proportion of FD Red-listed taxa (15%). The combined total for all non-fish vertebrates is only 7%, with all other taxonomic groups at similar percentage levels. For animals, only vertebrate taxa could be examined since to my knowledge there are no good overall data on invertebrates (but for an examination see Scudder 1996). Plants could not be included in this analytical comparison since the numbers summarized here and by Smith (1998) were for vertebrates only.

Protection

Forest Practices Code

The percentage of Forest Dependent fish taxa (Smith 1998) that receive particular protection under the FPC as Managed Identified Wildlife (MIW; B.C. MELP 1999*b*) is <1% (Fig. 3). This protection is for a single fish species, bull trout (*Salvelinus confluentus*). Only 2% of the total number of vertebrates listed as MIW are fish. Yet, 25% of all the vertebrate taxa officially designated as FD are fish. There are also more fish taxa listed as FD (69) than the total number of vertebrates listed as MIW (43).

Reptiles and amphibians are the only taxonomic categories that have more FD taxa listed as MIW under the Forest Practices Code than their proportional composition in the total number of vertebrates listed by the CDC (Cannings and Ptolemy 1998, Smith 1998, Cannings et al. 1999): 9% of all

CDC-listed vertebrates are reptiles and 2% are amphibians, yet the proportion of FD taxa listed as MIW is 100% for reptiles and 8% for amphibians.

Reptiles and amphibians also have more taxa designated as MIW than their proportions of the total FD vertebrate taxa. One percent of the total number of FD vertebrate taxa are reptiles and 5% are amphibians.

The other 2 non-fish vertebrate taxonomic groups, mammals and birds, also have high proportions of taxa designated as MIW in comparison to their proportions of the total FD vertebrate taxa. Mammalian taxa compose 22%, and bird taxa 47%, of all FD vertebrate taxa, yet 37% of mammals and 49% of birds are designated as MIW. Mammals also have somewhat more FD taxa listed as MIW under the FPC (26%) than their proportional composition of the total number of FD vertebrates (22%).

The combined non-fish vertebrate taxa compose 75% of FD vertebrate taxa, but 98% of all vertebrate taxa listed as MIW are non-fish vertebrates. Nevertheless, only 20% of all non-fish vertebrate taxa designated as FD are listed as MIW in the FPC. This percentage demonstrates another recurring theme, which is that no vertebrate groups except reptiles are receiving MIW protection under the FPC that is concomitant with their respective numbers of FD taxa. This situation and disproportion are worst in fish and in aquatic species and ecosystems.

Protected Areas Strategy

The Protected Areas Strategy for British Columbia incorporates populations of about 15% of Red-listed fish taxa at some level (Fig. 4). Only 9% of the Blue-listed fish taxa and 4% of the Forest Dependent fish taxa are in these areas. Fourteen percent of particular "hotspots" for fish and aquatic biodiversity (Haas 1998) are covered by protected areas at least in some way. Most of the incorporation of fish and aquatic biodiversity in these cases is incidental (Haas unpubl. data).

The PAS is based on a biogeoclimatic concept called ecosections (Demarchi 1995, also see Smith 1998, Land Use Coordination Office 1999). The ecosections are ecologically defined terrestrial regions with borders delineated by climate and physiographic features (Figs. 6–7). These ecosections can have limited correspondence to the distribution of fish and aquatic biodiversity in terms of their "hotspots," Red- and Blue-listed taxa (Fig. 4), or occupied drainages (Fig. 5). Many of the actual protected areas (Fig. 4) and the ecosections with more area protected (Fig. 6) are at higher

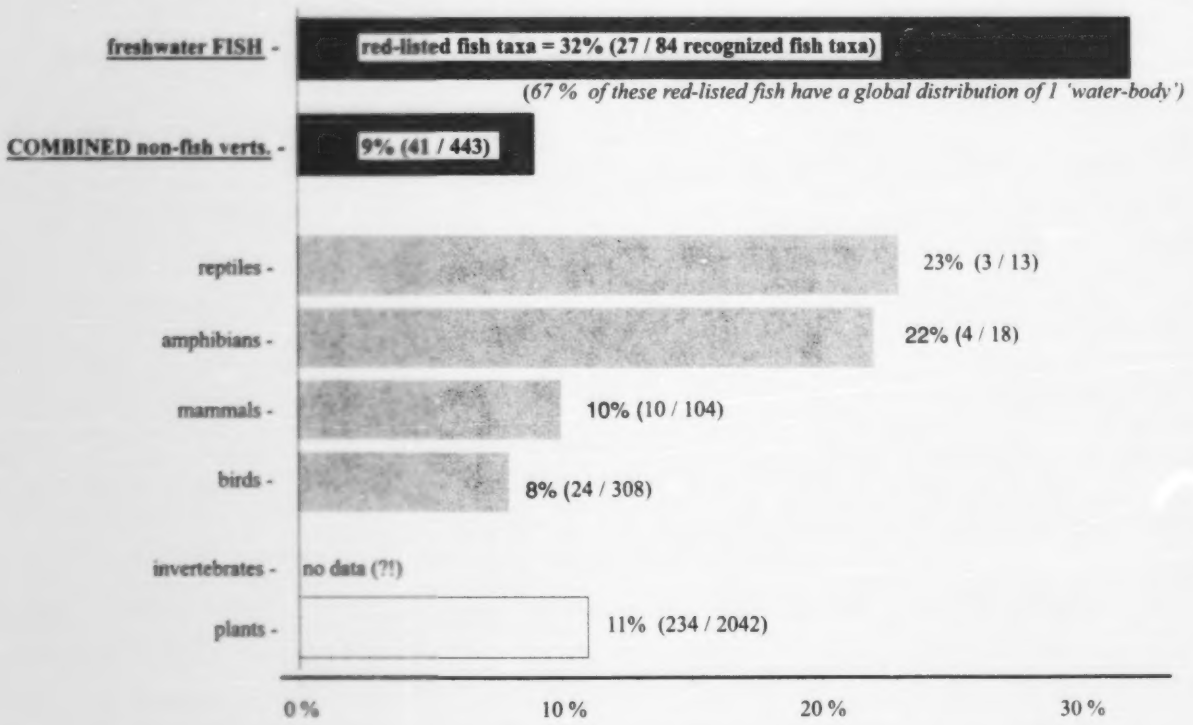

Figure 1. Percentages and numbers of Red-listed fish taxa compared to other non-fish vertebrate taxa and non-vertebrate taxa. Modified from Smith (1998). The term "taxa" is used as a surrogate for "species" due to incomplete taxonomies.

elevations (Fig. 7) than those at which many fish taxa occur. The relationship of the ecosections to the major overall drainage basins and their component watershed groups (B.C. MELP 1999*a*,*b*) is also not very strong (Fig. 5). This weak ecosection relationship is also the case with the post-glacial recolonizations and temporary drainage connections (Fig. 8) that structure most of the recent historical biodiversity and taxa in fish and in aquatic species and ecosystems.

DISCUSSION

Risks

The general public, and often even scientific, perception that fish and aquatic species and ecosystems are receiving enough, or even too much, of the overall conservation efforts and protection is not correct (McPhail 1993, Slaney et al. 1996, Haas 1998). Freshwater fish are at much more risk than any other taxonomic group designated and tracked by the CDC. These high risks are for Red-listed taxa (Fig. 1) and those designated as Forest Dependent (Figs. 1–2). The disproportionately higher risk for fish and for aquatic species and ecosystems is not matched by protection for them. Fish are inadequately represented under and within the Managed Identified Wildlife provision in the Forest Practices Code, which is skewed towards the other non-fish vertebrate taxonomic groups (Fig. 3). Fish and aquatic protection are also not well incorporated in the Protected Areas Strategy (Fig. 4), which has a similar terrestrial bias.

Protection

Forest Dependency and the Forest Practices Code

The FPC uses the Managed Identified Wildlife provision for "taxa that will not be adequately maintained through application of the Biodiversity and Riparian Management Area Guidebooks" (Forest Practices Code 1995, B.C. MELP 1999*b*). Forty-three vertebrate taxa presently have MIW status, while 282 vertebrate taxa are considered Forest Dependent (Smith 1998). This results in 15% of FD vertebrate taxa receiving MIW listing (Fig. 3). There would seem to be at least some shortfall in the MIW listings for vertebrates in general, even if some FD species are adequately protected by the FPC without MIW status. This situation is worst for fish, of which only 1 species, bull trout, is listed as MIW. Without accounting for commercial salmonids (e.g., Slaney et al. 1996), there are a minimum of 10 additional fish species that can be strongly argued as warranting special forestry consideration, and more fish taxa are likely candidates too (Haas 1998).

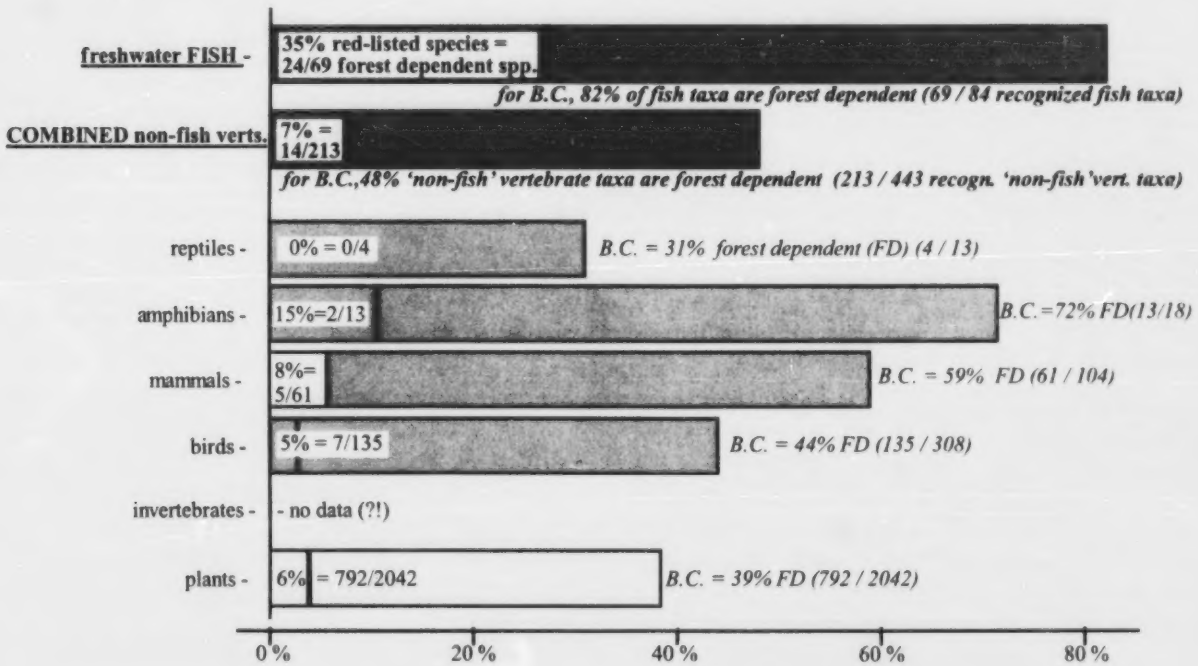
freshwater FISH -
35% red-listed species = 24/69 forest dependent spp.
for B.C., 82% of fish taxa are forest dependent (69 / 84 recognized fish taxa)
COMBINED non-fish verts.
7% = 14/213
for B.C.,48% 'non-fish' vertebrate taxa are forest dependent (213 / 443 recogn. 'non-fish'vert. taxa)
reptiles -
0% = 0/4
B.C. = 31% forest dependent (FD) (4 / 13)
amphibians -
15%=2/13
B.C.=72% FD(13/18)
mammals -
8%= 5/61
B.C. = 59% FD (61 / 104)
birds -
5% = 7/135
B.C. = 44% FD (135 / 308)
invertebrates -
- no data (?!)
plants -
6% = 792/2042
B.C. = 39% FD (792 / 2042)
0 %
20 %
40 %
60 %
80 %

Figure 2. Percentages and numbers of Forest Dependent fish taxa compared to other non-fish vertebrate taxa and non-vertebrate taxa. Modified from, and using forest dependency data from, Smith (1998). The term "taxa" is used as a surrogate for "species" due to incomplete taxonomies.

A difficulty under the FPC is that the listing of more fish taxa as MIW is likely to result in unacceptable impacts (fixed as >1 %) on the Annual Allowable Cut (AAC) in forest harvesting in British Columbia (B.C. MELP 1999*b*). An example of attempting to deal with this is the situation where 3 additional fish species were submitted for consideration as MIW with provisions for limiting impacts on the AAC worked into their accounts (Haas 1997*a-c*). These limiting provisions may be legitimate in accommodating both biological and economic considerations, but, as a result, all populations will not necessarily receive the best protection under the FPC that is generally accorded other MIW vertebrates. These 3, and likely other, fish taxa submissions are presently being reconsidered under the FPC (S. Rautio, B.C. Ministry of Environment, Lands and Parks, Wildlife Section, pers. comm.).

A further limitation is that the present mainstay of MIW protection is the establishment of Wildlife Habitat Areas (WHAs; Forest Practices Code 1995, B.C. MELP 1999*b*). These WHAs involve setting aside critical and often smaller habitat areas for specific life-history stages or important events such as breeding locations. However, the WHAs may not work well for aquatic systems and taxa since they do not account for the interconnectivity of streams, which occurs at least at some scale (e.g., Hartman and Scrivener 1990, Stanford and Ward 1993, Reid 1998). Preserving small areas in watersheds may simply be insufficient, since anything happening upstream in a watershed is probably destined to affect it downstream, and downstream impacts can also prevent access, etc. to upstream sites.

To be fair, the FPC upon establishment was apparently designed to protect salmonid fishes (Dr. A. Tautz, B.C. Ministry of Fisheries, Research Section, pers. comm.) through its inherent riparian provisions (Forest Practices Code 1995). At present, and based on this and other analyses and expert opinions in British Columbia and elsewhere (e.g., Coats and Miller 1981, Stanford and Ward 1992, Frissell and Bayles

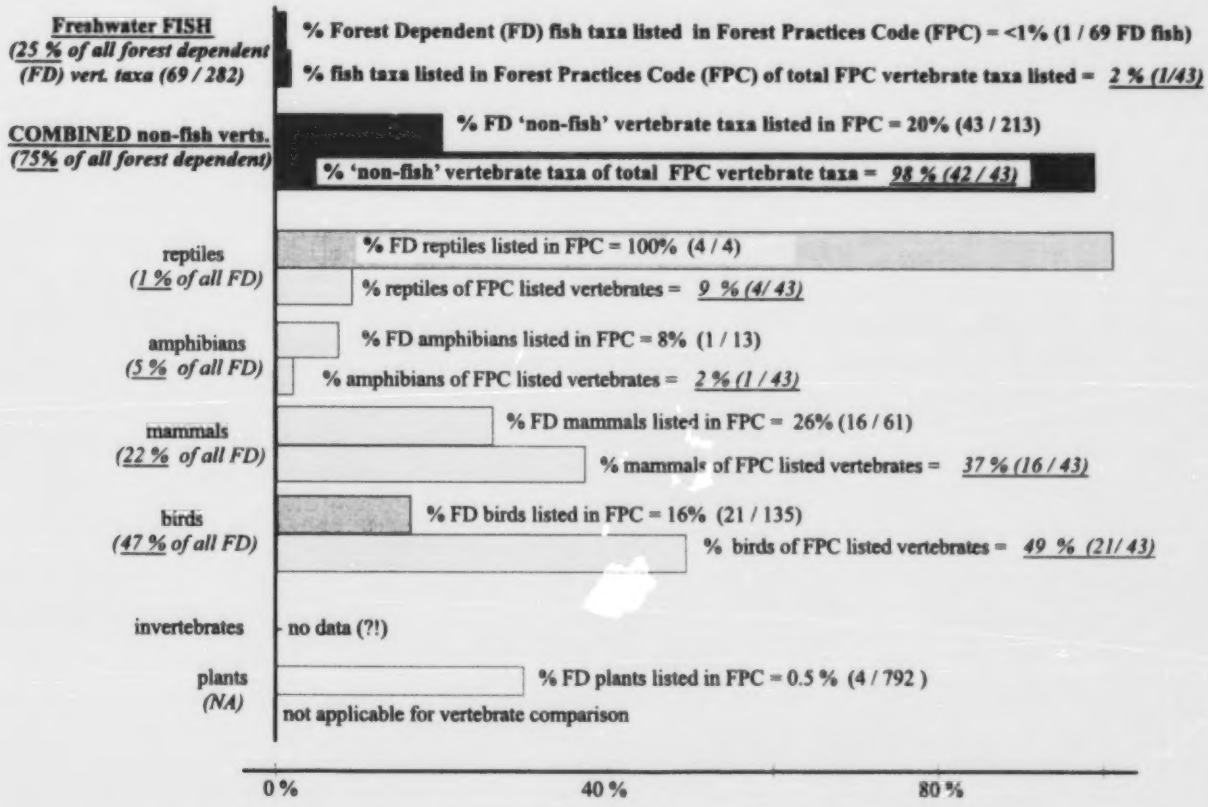
Freshwater FISH
(25 % of all forest dependent (FD) vert. taxa (69 / 282)
% Forest Dependent (FD) fish taxa listed in Forest Practices Code (FPC) = <1% (1 / 69 FD fish)
% fish taxa listed in Forest Practices Code (FPC) of total FPC vertebrate taxa listed = 2 % (1/43)
COMBINED non-fish verts.
(75% of all forest dependent)
% FD 'non-fish' vertebrate taxa listed in FPC = 20% (43 / 213)
% 'non-fish' vertebrate taxa of total FPC vertebrate taxa = 98 % (42 / 43)
reptiles
(1 % of all FD)
% FD reptiles listed in FPC = 100% (4 / 4)
% reptiles of FPC listed vertebrates = 9 % (4/ 43)
amphibians
(5 % of all FD)
% FD amphibians listed in FPC = 8% (1 / 13)
% amphibians of FPC listed vertebrates = 2 % (1 / 43)
mammals
(22 % of all FD)
% FD mammals listed in FPC = 26% (16 / 61)
% mammals of FPC listed vertebrates = 37 % (16 / 43)
birds
(47 % of all FD)
% FD birds listed in FPC = 16% (21 / 135)
% birds of FPC listed vertebrates = 49 % (21/ 43)
invertebrates
no data (?!)
plants
(NA)
% FD plants listed in FPC = 0.5 % (4 / 792)
not applicable for vertebrate comparison
0 %
40 %
80 %

Figure 3. The first row in each histogram pair shows the percentage and number of Forest Dependent (FD) taxa that are listed in the British Columbia Forest Practices Code (FPC). The second row compares the number of taxa in that category to the combined total number of vertebrate taxa (43) listed in the FPC, with the percentage of each category compared to all FD vertebrate taxa (282) on the y-axis. Data from Smith (1998).

1996), riparian protection alone does not appear to be fully working. Fish and aquatic species and ecosystems need further, specific consideration. Outside of the often limited assistance of MIW provisions, the only present alternative options under the FPC are "general wildlife measures" (GWMs) outside of WHAs (B.C. MELP 1999*b*) and "higher level planning" within Land and Resource Management Plans (LRMPs; Land Use Coordination Office 1993, 1997, 1999; B.C. Ministry of Forests 1996).

GWMs have not yet been used for any MIW taxa. There again are concerns about their likely higher impact on the AAC. GWMs can partially or entirely limit any forest practice as defined under the FPC. This includes such things as road construction, road maintenance, grazing, haying, and timber harvesting, which are categorized under the following headings: access, range, recreation, restoration and enhancement, and silviculture. Once GWMs are approved, they, too, are legally binding.

LRMPs have been worked on in many regions of British Columbia, and provincial level strategies have even been enacted for such large-ranging, charismatic species as grizzly bears (*Ursus arctos horribilis*; B.C. MELP 1995). LRMPs attempt to consider all resource values and require public participation, interagency coordination, and consensus-based land and resource management decisions. LRMPs cover subregional areas of approximately 15,000–25,000 square kilometres. The plans try to establish direction for land use and specify resource management objectives and strategies. They work toward a comprehensive, accepted, and approved management framework to guide resource development and more detailed planning. In general, they have usually not yet fully addressed fish and aquatic species and systems. Future use of higher level planning within LRMPs to help protect these should be examined. Such procedures may be a key component for aquatic ecosystem protection given the limitations of, and the 1% impact cap on the AAC by MIW provisions.

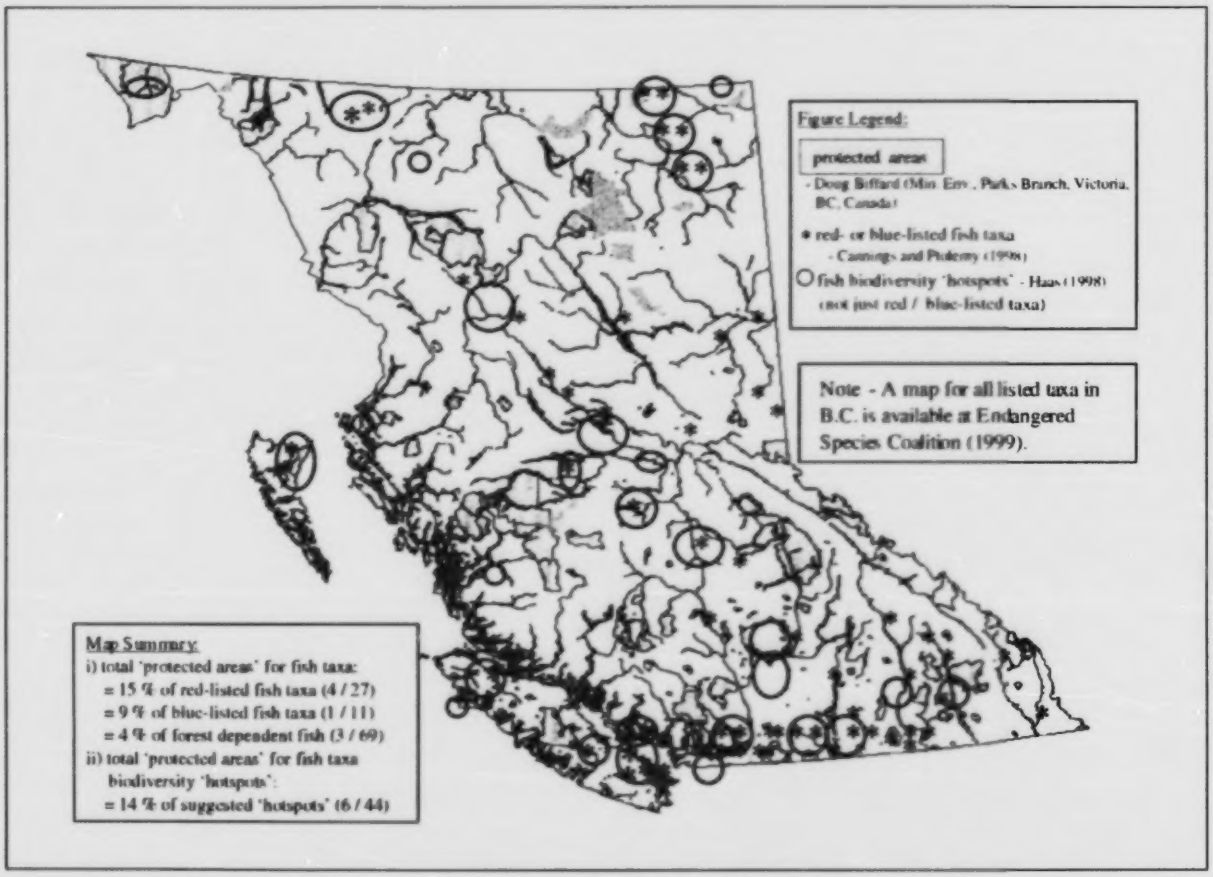
Figure Legend:
protected areas
- Doug Biffard (Min. Env., Parks Branch, Victoria, BC, Canada)
* red- or blue-listed fish taxa
- Cannings and Ptolemy (1998)
O fish biodiversity 'hotspots' - Haas (1998)
(not just red / blue-listed taxa)
Note - A map for all listed taxa in B.C. is available at Endangered Species Coalition (1999).
Map Summary:
i) total 'protected areas' for fish taxa:
= 15 % of red-listed fish taxa (4 / 27)
= 9 % of blue-listed fish taxa (1 / 11)
= 4 % of forest dependent fish (3 / 69)
ii) total 'protected areas' for fish taxa biodiversity 'hotspots':
= 14 % of suggested 'hotspots' (6 / 44)

Figure 4. The British Columbia Protected Areas Strategy in relation to the distribution of Red- and Blue-listed fish taxa, and fish biodiversity hotspots.

Furthermore, such higher level planning might work beyond the 12% protected areas and encourage necessary conservation within the entire province, its working economy, and even non-endangered or non-listed taxa (in British Columbia, often termed "yellow-listed" taxa).

Protected Areas Strategy

The key objective of the Protected Areas Strategy in British Columbia is to increase representation and protection of ecosections (Land Use Coordination Office 1999). This means the preservation of enough of all ecosystem types of distinct biogeoclimatic zones that are composed of geological, climate, and vegetation features (Fig. 6). By doing this, it is anticipated that most species utilizing these distinct habitat regions will be coincidentally protected (e.g., Smith 1998). However, many ecoprovinces and ecosections can have little relation to important features of aquatic ecosystem and species (Fig. 5; Frissell 1993, Hitt and Frissell 1999, Newall and Magnuson 1999). What is important to fish and aquatic systems are overall drainage basins, distinct watershed groupings, isolated populations, aquatic habitat, and fish taxa distributions and composition, both at present and historically (e.g., McPhail and Carveth 1993*a*, Haas 1998). These important "aquatic ecosections" are not necessarily represented by the terrestrial-based biogeoclimatic ecosection or zone concept (Figs. 5–7).

Habitat protection for fish biodiversity might work in some cases within "complete" protected areas, given that 18 of the 27 Red-listed freshwater fish taxa (Cannings and Ptolemy 1998) have a global distribution of a single British Columbia site (Haas 1998). This means that 67% of British Columbia's Red-listed freshwater fish taxa are found globally in only 1 water body in British Columbia (Fig. 4; Endangered Species Coalition 1999), although several of these water bodies are quite large for complete protection (Cannings and Ptolemy 1998, Haas 1998). As well, 9 of the 38 Red- or Blue-listed taxa are peripheral isolates occurring in disjunct distributions or at the edge of the range for that taxon (Haas 1998). This means that 24% of these listed fish taxa are found in British Columbia in only 1 water body, but that these species have a larger distribution outside of the province. Peripheral isolates can nonetheless be interesting and important since they may be subject to distinct evolutionary or ecological forces, because they are somewhat or wholly separated from the rest of their taxon and are at the limits of their range (Scudder 1989, McPhail and Carveth 1993*a*, Haas 1998). Also, if the jurisdictional, and not just biological, range of a species is important, then these peripheral isolate fish taxa do occur in only 1 water body in British Columbia. As a combined total, 71% of the Red- and Blue-listed freshwater fish taxa in British Columbia are only found in 1 water body here.

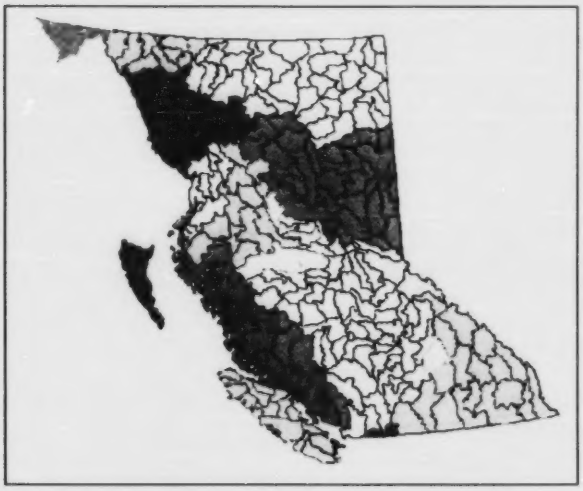

Figure 5. British Columbia's 246 distinct watershed groups (B.C. MELP 1999*a*,*b*), with their major overall drainage basins highlighted.

Such single-site patterns could also receive alternative protection under the Ecological Reserve System in British Columbia (B.C. MELP 1999*a*), but so far only 2 of all the ecological reserves (2 of approx 136, or 1.5%) were specifically set aside for fish. A difficulty with protecting fish and aquatic species and ecosystems in ecological reserves is that watersheds are interconnected. Preserving small areas in watersheds may be insufficient, and this must be anticipated in the planning of ecological reserves and protected areas, and assessment of resource utilization impacts (e.g., Haas 1998). The unique fish taxa found in some smaller isolated lakes in British Columbia may be the best candidates for preservation under such protection systems. The danger that remains in even these cases is the introduction of exotic fish species that can and have wiped out entire fish taxa (Cannings and Ptolemy 1998, Haas 1998). This threat of unauthorized exotic species introductions must be considered (McPhail and Carveth 1993*a*, Ricciardi and Rasmussen 1998), but admittedly is likely hard to address.

Another aspect of many protected areas is that they were previously set aside for scenic vistas and are often located at higher altitudes (Figs. 4, 6, and 7). These protected areas were created more for recreation than conservation. This lack of utility of protected areas for conservation is particularly true for biodiversity in fish and aquatic species and ecosystems, which are generally best represented at lower elevations in larger drainages (McPhail and Carveth 1993*a*,*b*; Sparks 1995; Haas 1998; Newall and Magnuson 1999). These lower elevation habitats have received little protection and the most human impact. Much of this land is in the valleys formed by water drainages, and is often already utilized, making it unsuitable or difficult for preservation.

In many instances, traditional parks also had increased access, use, and pressure on the ecosystems they were, in theory, protecting. Again, this is particularly true for fish and for aquatic species and ecosystems, since angling and water use are still permitted in most protected areas in British Columbia, in spite of this not being the case for other taxonomic groups of organisms (e.g., collection of plants and hunting are often prohibited). Furthermore, fish from outside the protected area's drainage were, and still are, stocked into their water bodies, in some cases even introducing exotic fish species not native to British Columbia. Fish stocking in watersheds, with portions fishless or not, also has had considerable impacts on aquatic taxa and their ecosystems (e.g., Dill and Cordone 1997). Many national parks in British Columbia (Parks Canada 1999) also had their own angling regulations under federal jurisdiction and these often were more lax. A final and key consideration is that the laudable goal of the PAS is to preserve 12% of British Columbia's land base. With 10–11% already set aside (Smith 1998), and with disproportionately very few fish protected, this strategy should consider immediate and strong aquatic representation.

Other Related Important Issues

Taxonomy and Identification

Another key unaddressed conservation issue is taxonomy. The term "taxa" is used instead of "species" by the CDC since the number of taxa it recognizes is its legitimate best estimate (but not necessarily universally accepted). For instance, Haas (1998) lists 6 other fish taxa/species not presently recognized and at least 10 more taxonomic problems within recognized taxa that may further increase the number of fish species in British Columbia. Emphasis of the scope and state of this taxonomic problem is that 25% of the fish taxa recognized by the CDC do not even have proper taxonomic nomenclature. All of these un-named fish are also either Red-listed, extinct (Cannings and Ptolemy 1998), or

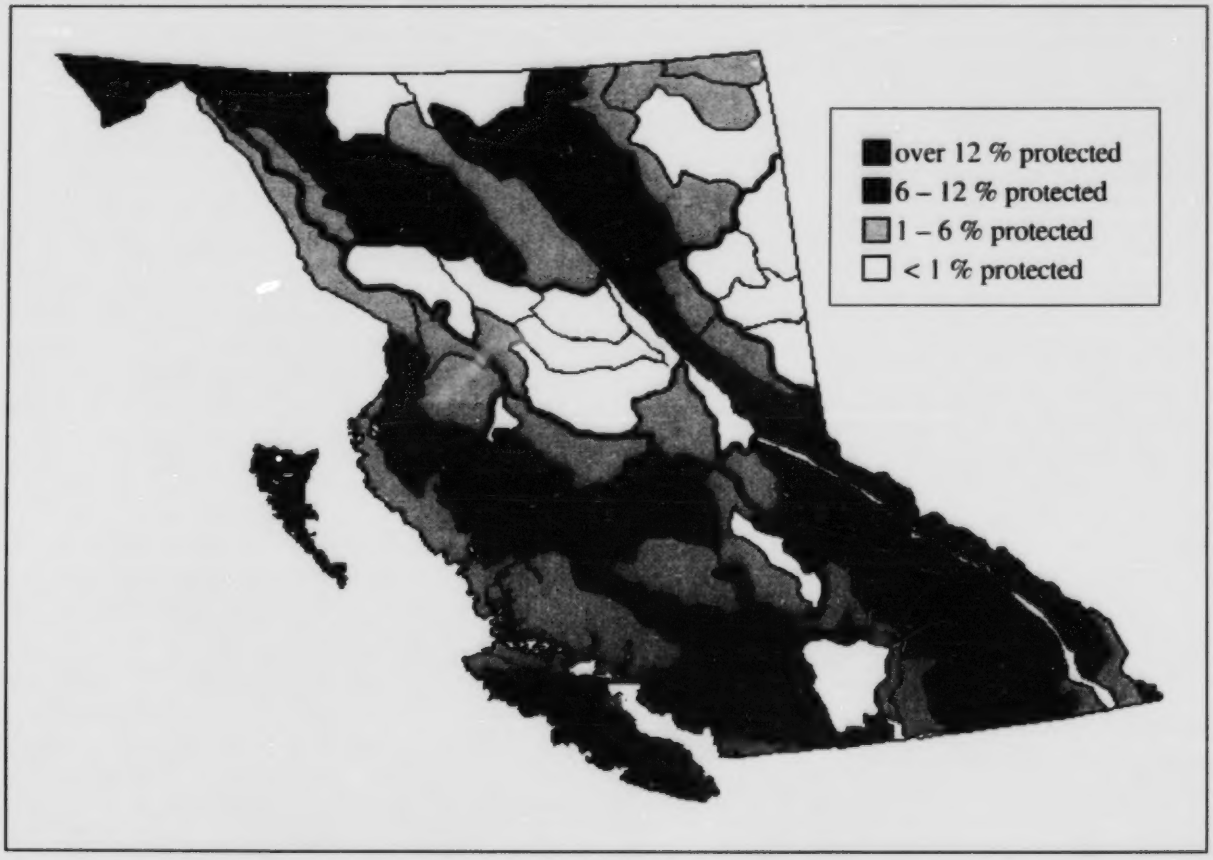

Figure 6. British Columbia's ecoprovinces (heavy black lines), ecosections (regular black lines), and percentage in protected areas (see legend) in each ecosection. Ecoprovince and ecosection data are from Demarchi (1995). Protected area percentage data are from Smith (1998).

prioritized as most at risk (Haas 1998). Conversely, several fish taxa with specific nomenclature recognized by the CDC are not necessarily accepted as distinct species by all fish biologists (e.g., McPhail and Carveth 1993*a*,*b*; Haas 1998) or the American Fisheries Society (Robins et al. 1991), which is the generally accepted governing body for fisheries in North America. A last testament to this issue is that the bull trout is the only fish species recognized under the FPC, and even it was not recognized as a distinct species in British Columbia until this decade (Haas and McPhail 1991), with its resurrection as a species occurring only slightly earlier (Cavender 1978).

These taxonomic problems are exacerbated by an average species misidentification rate of at least 24% (range 0–100% for different fish taxa) in fisheries inventory work (Resources Inventory Committee 1997*a*,*b*, 1998) for the FPC (Haas et al. submitted). This figure is also only for described, named, and generally accepted fish species. Knowledge of ecology and life-history for most fish and aquatic taxa is in even worse condition than knowledge of their taxonomy and species identification (Murphy 1990, McPhail and Carveth 1993*a*, Haas 1998). Correct species identification is the fundamental basis of biological work (e.g., Altaba 1997, Pimm and Lawton 1998, Haas et al. submitted), and proper species management and protection requires good working ecological knowledge based on good identification (e.g., Edwards and Morse 1995, Barbosa and Galdean 1997, Kareiva et al. 1999). A strongly related issue is that proper identification and work on biodiversity depends on natural history museums and their personnel, which presently do not receive this acknowledgement or adequate support in British Columbia and the rest of Canada (Haas et al. submitted).

Unique Within-species Variability and Patterns

As already discussed, aquatic species and ecosystems and taxa are best represented by drainage and watershed

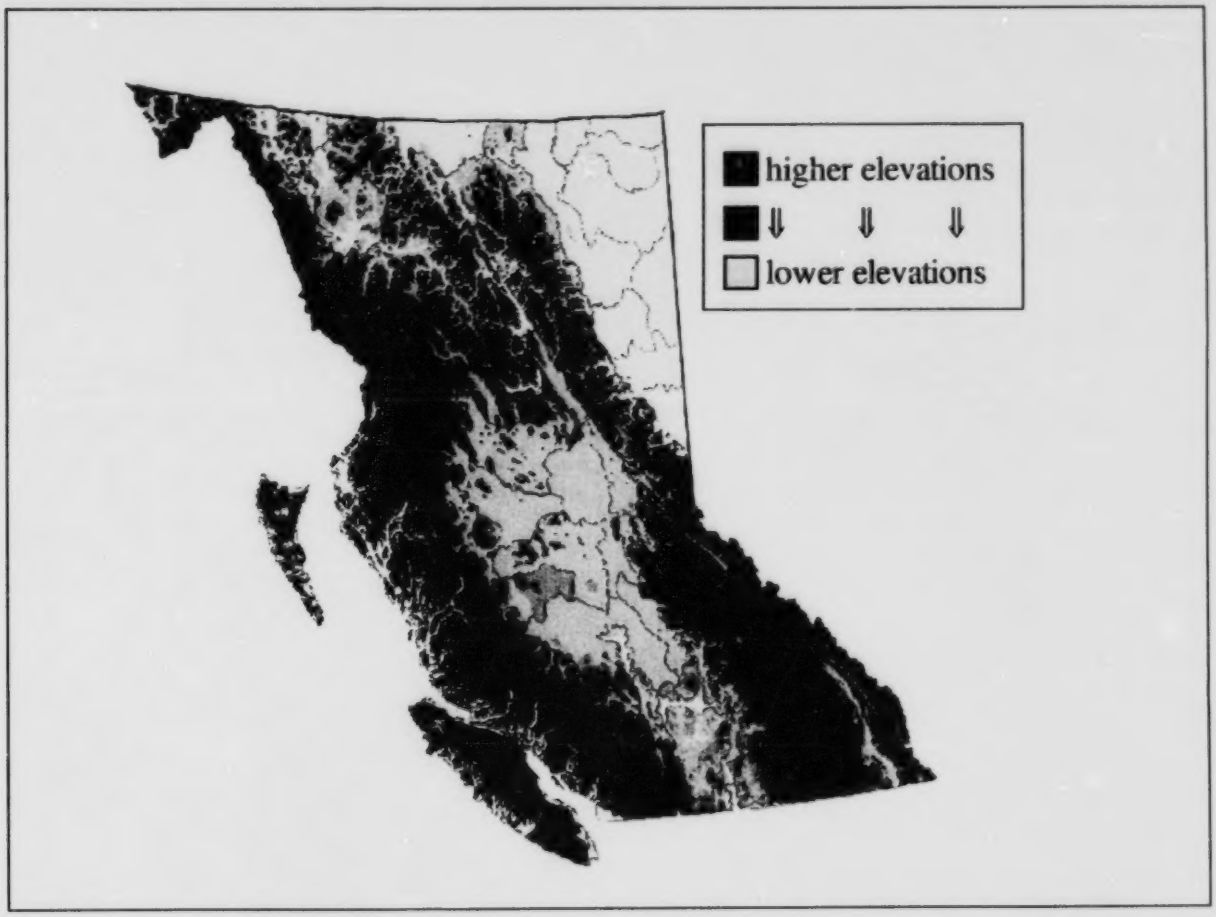

Figure 7. British Columbia topographical map showing elevation in relation to the ecosections in Figure 6.

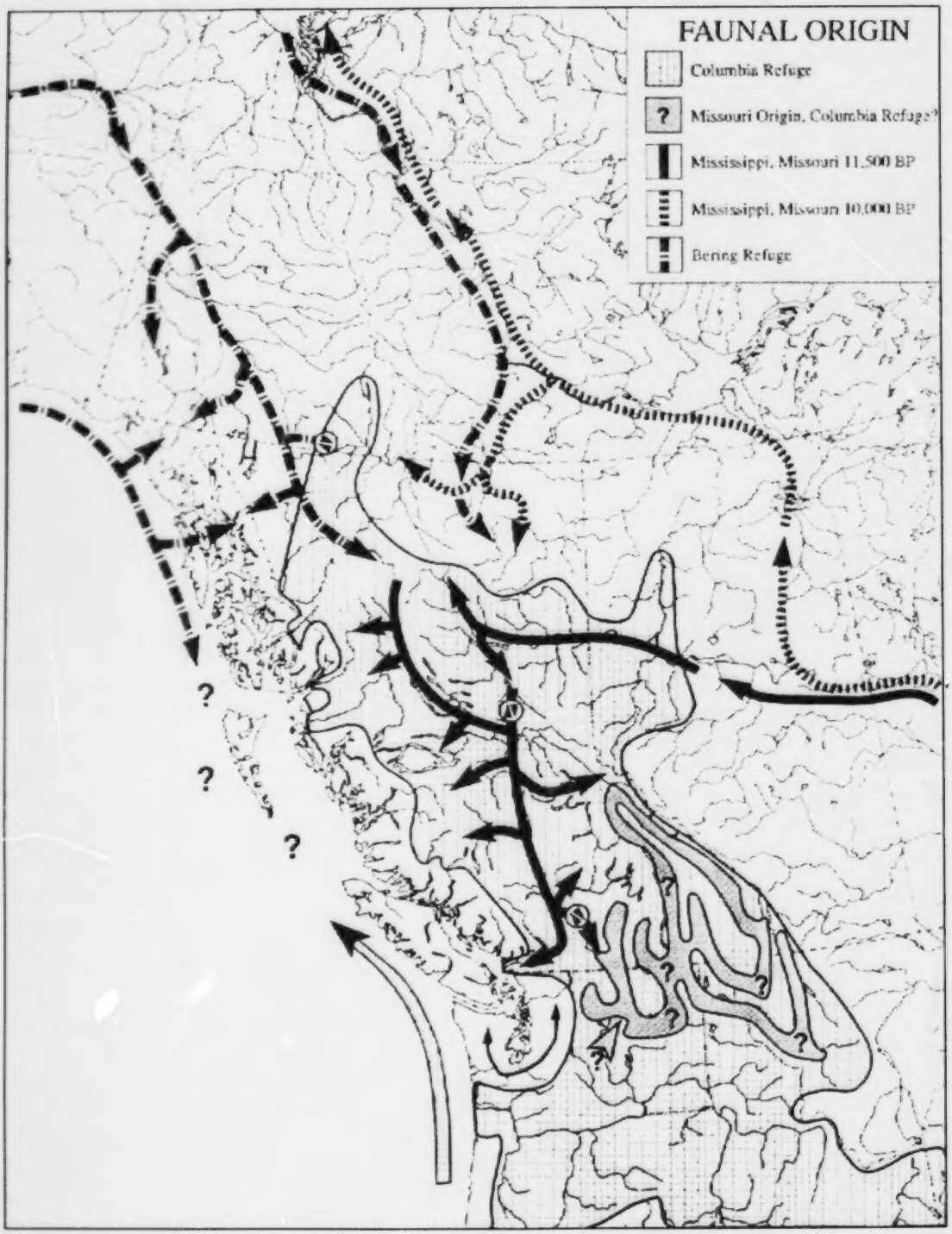

Figure 8. Fish glacial refugia, postglacial recolonization, and present fish distributions based on them. Map is from McPhail and Carveth (1993*b*).

patterns. These aquatic distribution patterns also produce related, and stronger, within-species variability (e.g., Taylor and Haas 1996) than for most other vertebrates, particularly than for those that are warm-blooded (e.g., Flather et al. 1998). Aquatic species and ecosystems are isolated from each other (Fig. 5) by being in separate overall drainages (e.g., Columbia vs. Fraser rivers) and, within those, by being in distinct watershed groups (e.g., Columbia → Kootenay → Elk River →). Even within the watershed groups, smaller aquatic ecosystems are still further isolated by being above barriers such as waterfalls. These populations may also have been strongly affected by other events such as founder effects. This all means that freshwater and anadromous fish have a tremendous level of within-species diversity even though the actual number of recognized fish taxa in British Columbia may not be as high as for other non-fish vertebrate taxonomic groups (e.g., Douglas et al. 1998*b*). The freshwater fish fauna of British Columbia is not rich in species, but does have a unique provincial make-up and tremendous within-species variability (McPhail 1993, McPhail and Carveth 1993*a*, Haas 1998). These fish species are also often not naturally found anywhere else in Canada or, in some cases, anywhere else in the world, whereas many fish in other parts of Canada are distributed in more than 1 province.

In conjunction with such unique aquatic distribution features, British Columbia's geologically recent glacial history has also had dramatic impacts on freshwater fish and other aquatic species biodiversity (McPhail and Lindsey 1970, 1986; Lindsey and McPhail 1986; McPhail and Carveth 1993*a*; Haas 1998). During the last Wisconsin glaciation, British Columbia was almost completely covered with ice for 40,000 years, up until approximately 10–15,000 years ago, when the glaciers began to recede (Fig. 8). During this glacial epoch, fish were wiped out in British Columbia or they survived glaciation in a limited number of distinct glacial refugia beyond the reach of the ice-sheets. The glacial refugia were well separated and had quite different environments, with the fish taxa surviving in them having had 40,000 years to evolve into somewhat or completely distinct forms. The vast majority of often already differentiated fish taxa then recolonized British Columbia from 1 or more of these distinct glacial refugia, often through temporary post-glacial drainage connections that no longer exist (Fig. 8).

After the fish recolonized the province, they had 10–15,000 years in which to further evolve in distinct drainage basins and watershed groups, again often with quite different environments, species compositions, etc. Fish taxa in British Columbia thus have within-species variability resulting from both historical glacial refugia and more recent ecological environments (Haas 1998, Haas and McPhail submitted). These types of within-species variability are much less prevalent in many terrestrial, and particularly warm-blooded, organism groups, and often need not be considered in these cases. Such terrestrial populations tend to be less isolated and more mobile, at least on a smaller scale. Those non-fish or terrestrial organisms that are highly isolated on a larger scale are also often given formal taxonomic status as subspecies or geographic isolates (e.g., O'Brien and Mayr 1991, Rojas 1992). This is again not the case for fish.

The recognition of within-species variability is well established and is explicitly considered in commercial anadromous salmonids as the stock concept. In this concept, populations of particular anadromous salmonid fish taxa are recognized as stocks somewhat or totally distinct from other populations of the same species, due to geographic, temporal, or ecological isolation. This concept even has legislative status under the United States Endangered Species Act, in which distinct salmonid stocks are recognized as "Evolutionary Significant Units" that can each be separately designated as threatened or endangered (Waples 1995). A problem in British Columbia is that only 57% of the estimated 10,000 salmonid stocks could be assessed due to a lack of information. It is further likely that the smaller stocks compose much of the overall salmonid biodiversity, and those are less well known (Slaney et al. 1996). Furthermore, even within the 5,700 stocks that could be assessed, there still were 142 extinct, 624 at high risk, 78 at moderate risk, and 230 of special concern (Slaney et al. 1996). None of these or other salmonid stocks are listed by the Conservation Data Centre and most do not receive other special protection or attention (Haas 1998).

Non-commercial or Non-sport Fish Taxa

Although such large problems exist for major commercial and sport fish, the situation is even worse for non-commercial or non-sport fish taxa (and for other non-fish aquatic taxa) in British Columbia (Haas 1998, unpubl. data) and elsewhere (e.g., Douglas et al. 1989, 1998*b*; Holden 1991). The stock concept and process is legitimate and useful, but it has not been extended to include most British Columbia freshwater fish taxa. This is in spite of these other, mostly non-salmonid fishes being largely less saltwater-tolerant and less migratory, and thus likely more structured and isolated as discrete populations (Haas 1998, Haas and McPhail submitted). To give an example for some perspective, coastrange sculpins (*Cottus aleuticus*) are often found in the same sites as coastal commercial and sport species of salmonids, for which some 2,000 stocks are estimated (Slaney et al. 1996). However, the coastrange sculpin, and other non-commercial or non-sport fish taxa, are treated at the species level, if they receive consideration at all. There is little, if any, recognition of their within-species differentiation (e.g., Douglas et al. 1989, 1998*b*), in spite of the aforementioned evidence suggesting it could be stronger than for anadromous migratory salmonids.

Much of the fish biodiversity in British Columbia is likely

represented in the taxa and their within-species variability that is presently unrecognized and unaccounted for in the province's predominantly non-salmonid fish (e.g., Haas and Ritchie in press). These non-salmonid fish taxa also compose most of the Red- and Blue-listed fish in British Columbia. Such conservation bias toward economically important species is largely unacceptable for other non-fish vertebrate taxonomic groups (Cannings et al. 1999, Douglas et al. 1998*a*, Harper et al. 1999). This situation is even worse for the other totally unrecognized taxonomic organisms and groups present in aquatic ecosystems such as invertebrates (Scudder 1996).

MANAGEMENT AND CONSERVATION IMPLICATIONS

Freshwater fish and aquatic species and ecosystems in British Columbia have the most Red-listed taxa, and many of these are found only in (or have disjunct distributions in) a single water body, often globally. They have the most taxa designated as Forest Dependent of any vertebrate taxonomic groups. Their disproportionately higher risk is not matched by protection or legislation. They also are inadequately represented under the Protected Areas Strategy. They should receive consideration now, given that the designated objective of 12% protected areas is soon to be reached (Smith 1998). Protected areas must consider conservation and not just recreation.

Fish and aquatic species and ecosystems are not equally represented in the Forest Practices Code or its riparian provisions. At least in specific instances of isolated single taxa occurrences in smaller water bodies, fish warrant more direct status and full protection as Managed Identified Wildlife. However, in many cases it is likely insufficient to only set aside particular smaller aquatic locations such as occurs under Wildlife Habitat Areas or Ecological Reserves. Protection under both the PAS and the FPC should recognize that watersheds are physically integrated ecosystems. This could be addressed with General Wildlife Measures under the FPC and through FPC higher level planning procedures such as Land Resource Management Plans. However, acceptable impacts (1%) on the Annual Allowable Cut could likely only be somewhat mitigated through LRMPs.

Fair and rigorous criteria that encompass all taxonomic groups in proper perspective should be implemented (Meffe et al. 1998, Colyvan et al. 1999, Kareiva et al. 1999). Terrestrial and aquatic biologists must work together as overall conservation advocates and not just on their own agendas. In some cases, the real economic values of biodiversity and conservation (e.g., Boyle and Bishop 1987, Stevens et al. 1991, Gavaris 1996, Loomis and White 1996, Baker and Pierce 1997, Bookbinder et al. 1998) could also be considered within this framework.

An aquatic ecosection system should be investigated and delineated (Moyle and Ellison 1991, Hitt and Frissell 1999, Newall and Magnuson 1999). The PAS in British Columbia is presently based on ecoprovinces and ecosections that utilize largely terrestrial criteria. The ideal would be integration of aquatic ecosections with terrestrial ones (e.g., Haas and McPhail submitted). At a minimum, distinct aquatic features and linkages must receive more consideration under habitat protection plans. Some effort towards an aquatic ecozone classification has been attempted in British Columbia (Perrin and Blythe 1998, also see Emmons et al. 1999), but it has not been validated for its correspondence to the actual biological distributions of aquatic taxa (e.g., see lack of correspondence in Wright et al. 1998).

There are particularly significant species-level problems or deficiencies for fish and aquatic species with regards to taxonomy, identification, life-history, and ecology (e.g., Collette and Vecchione 1995). There is still a strong need for fundamental research (e.g., Efford 1994, Mosquin et al. 1995, Auditor General 1998, Balmford and Gaston 1999), particularly where species nomenclature, identification, and knowledge are critical, such as under the FPC or within any upcoming federal endangered species legislation (Wildlife Habitat Canada 1995, Auditor General 1998, Spurgeon 1999).

Fish and aquatic biodiversity is more uniquely structured within species, and has both historical and ecological components. This is unrecognized for all but some commercial or sport salmonid fishes, and their conservation has not yet been addressed by the FPC or the CDC. Even for these salmonids, knowledge and management is biased towards larger populations, with many smaller stocks being poorly understood and unmanaged.

All aquatic taxa and stocks, ultimately including marine fish and ecosystems (Zacharias and Howes 1998), should be directly taken into account for protection (e.g., Franklin 1993, Orians 1993, Kendall 1999, Wardle 1999). This is already the case for non-commercial, not just commercial, taxa in other non-aquatic taxonomic groups such as birds and mammals (Haas and Ritchie in press). It should also be considered that effective conservation must work across British Columbia, its working economy, and even for non-endangered or non-listed taxa in all groups.

The poor fish and aquatic background information available warrants consideration of a risk avoidance and adaptive approach to conservation and management. Caution in favour of conservation is needed and it should be related to biological uncertainties and impact time frames (e.g., Kareiva et al. 1999). Conservation should, nonetheless, start now and work with "known data" in spite of quantitative information gaps. Stricter biological requirements could be implemented in the interim to help deal with this, and their effectiveness could be monitored and should be accommodated when potential impacts are unknown, large, or long-

term. Many fish taxa require immediate, and also fairly obvious (at least on a gross or fundamental [e.g. Naiman et al. 1992] level), conservation measures (e.g., Murphy 1990). There are recommendations and prioritizations for conservation, research, and data needs for British Columbia fish and for aquatic species and ecosystems in McPhail and Carveth (1993*a*) and Haas (1998).

ACKNOWLEDGEMENTS

Sincere thanks to B.C. Conservation Data Centre personnel, conference organizers/workers, D. Biffard, C. Ford, S. McAdam, H. Millen, Dr. R. Smith (and Co.), Dr. A. Tautz, and B. Woodbridge. Other supportive individuals were D. Atagi, L. Darling, Dr. T. Down, A. Haas, J. McPhail, B. Muttley, F. Muttley, M. Nevin-Haas, M. Porter, K. Smallwood, and N. Tallon. My conference expenses were covered by a research grant from Forest Renewal British Columbia. The analyses and interpretations are mine, and are not implied to reflect official provincial government policy. This paper is meant as a hopeful call for attention and action, and not at all to point blame or to imply other taxonomic groups and ecosystems are less worthy of conservation.

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A Review of Salmon as Keystone Species and Their Utility as Critical Indicators of Regional Biodiversity and Ecosystem Integrity

K. D. Hyatt
Canada Department of Fisheries and Oceans
Science Branch, Stock Assessment Division
Pacific Biological Station, Nanaimo, BC, V9R 5K6, Canada
HyattK@pac.dfo-mpo.gc.ca

L. Godbout
Canada Department of Fisheries and Oceans
Science Branch, Stock Assessment Division
Pacific Biological Station, Nanaimo, BC, V9R 5K6, Canada

ABSTRACT

Seven species of anadromous salmon comprised of more than 9000 local populations return annually to thousands of streams in coastal and interior watersheds throughout British Columbia. Salmon are commonly used as biological indicators of state changes in marine or aquatic environments because they are: (i) relatively widespread in freshwater and marine ecosystems, (ii) highly valued from economic and cultural perspectives, (iii) reasonably diverse in terms of species, life histories and genetic variations within species, (iv) highly sensitive to environmental change at each of several life history stages, and (v) commonly the subjects of long term assessment programs. Observations from studies of salmon allow us to draw inferences about changes in the state of aquatic and riparian ecosystems on a variety of temporal scales ranging from weeks to centuries and spatial scales ranging from meters to thousands of kilometers. An increasing body of research has confirmed that freshwater and especially anadromous salmon serve as keystone species that exert control as predators, prey and as suppliers of critical nutrients in both aquatic and terrestrial riparian ecosystems. Important ecosystem linkages to salmon exist at levels ranging from primary producers, through aquatic invertebrates, to aquatic vertebrates, to terrestrial invertebrates and finally to a diversity of terrestrial vertebrates including large carnivores, small mammals, amphibians and even birds. Given their demonstrated role as keystone species, their importance as west coast cultural icons for people of all races and their long history of careful assessment as an economically important regional resource, salmon exhibit utility for continued development as a highly cost effective indicator of the functional integrity of aquatic and terrestrial riparian ecosystems and of their continued potential to support both species populations at risk (bears, eagles, steelhead etc.) or general levels of biodiversity throughout much of British Columbia.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Provisional Evaluation of the Status of Westslope Cutthroat Trout in Canada

David W. Mayhood
Freshwater Research Limited
1213 - 20 Street N.W., Calgary, AB, T2N 2K5, Canada
dmayhood@fwresearch.ab.ca

ABSTRACT

I surveyed existing publications, reports, historical documents, and selected agency files, conducted field surveys, and interviewed reliable observers knowledgeable about selected waters to determine the native and present distribution of westslope cutthroat trout (*Oncorhynchus clarki lewisi*) in Canada. I evaluated this information in the light of existing knowledge of potential threats to arrive at provisional determinations of risk status. In British Columbia, westslope cutthroats once occupied: the mainstem of the upper Kootenay River and its major tributaries below impassable barriers; certain Columbia River headwaters; and some small, disjunct headwaters, mostly above barrier falls in the lower Kootenay, Columbia, and South Thompson drainages. Most of the apparently genetically pure native stocks still extant in several upper Kootenay tributaries are vulnerable to hybridization with, or replacement by, accessible nonnative salmonids now widespread in that drainage. The population and genetic status of most British Columbia stocks, including many introduced populations, remains undocumented. In Alberta, westslope cutthroats once occupied the mainstems of the Bow and Oldman rivers and their major tributaries below impassable barriers, downstream far into the plains. Except in the upper Oldman and Castle river basins, native stocks are now rare in Alberta, existing only in short reaches of a few isolated headwaters. Genetically pure native stocks are endangered due to their limited and highly fragmented distribution, continued exposure to introgressive hybridization, probable competition from introduced salmonids, continued habitat damage and loss, inadequate legislation and enforcement, and continued exposure to overfishing of some small populations. Already-introgressed stocks are more common, but are threatened for the same reasons. The numerous introduced populations are vulnerable. All but a few are small, isolated, originate from a single stock, and may lack the genetic resources to adapt to future environmental change. Population and genetic surveys are urgently required throughout the Canadian range.

Key words: Alberta, Columbia River, Kootenay River, *Oncorhynchus clarki lewisi*, westslope cutthroat trout.

The present wave of human-induced extinction of plants and animals, comparable in magnitude to the great natural extinctions in the fossil record (Reid and Miller 1989) has not spared the fishes of North America (Ono et al. 1983). Three genera, 27 species, and 13 subspecies have gone extinct in the last century on this continent (Miller et al. 1989). As of 1990, 292 fish species, or 28% of the North American fauna in fresh water, were in various categories of risk: indeterminate, rare, vulnerable, endangered, or extinct (Williams and Miller 1990). A total of 364 North American freshwater fishes are deemed to warrant protection because of their rarity (Williams et al. 1989). Presently in Canada, 57 species, subspecies, and special populations of freshwater fishes are officially recognized as either in some category of risk or have already been declared extinct (Campbell 1997).

These statistics touch only on the surface of the problem. Many North American fishes at risk are not yet officially recognized as such. The westslope cutthroat trout (*Oncorhynchus clarki lewisi*) is an example (Behnke 1992, American Wildlands et al. 1998). This subspecies—native to southeastern British Columbia, southwestern Alberta, western Montana, northern Idaho, and small parts of Wyoming, Oregon, and Washington—has only recently been proposed for listing under the United States Endangered Species Act. According to extensive evidence cited by the petitioners (American Wildlands et al. 1998), westslope cutthroats occupy just 19% of their native range, are represented by viable populations in 8% of the native range, and are abundant in just 2% of that range in Montana. In Idaho, viable populations remain in 36% of the native range, and are strong in just 11% of that range. Native populations on the periphery

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

of the native distribution in Wyoming, Oregon, and Washington have been similarly devastated. Most U.S. stocks have hybridized introgressively with other black-spotted trouts, especially rainbow trout (*O. mykiss*) and Yellowstone cutthroat trout (*O. c. bouvieri*).

As yet there has been no comprehensive assessment of the status of this fish in Canada. Recently, cutthroat trout were recognized as being at high risk with regard to forestry considerations in British Columbia, and research on the species was recommended as a high priority (Haas 1998). The westslope subspecies is not included on the Red or Blue lists in British Columbia (Cannings and Ptolemy 1998) or Alberta, and is not presently being considered for listing in Canada (Campbell 1997). I assessed the status of westslope cutthroat trout in parts of Alberta and British Columbia (Mayhood 1995, Mayhood et al. 1997*a*, 1998), but these analyses considered only parts of the range, or considered the whole range at a scale too coarse to support a reasonable overall assessment of conservation status. This paper summarizes the status of westslope cutthroat trout within its native range in this country. The assessment is provisional until up-to-date population surveys, and surveys of the true extent of hybridization using reliable biochemical genetic techniques, are completed throughout the Canadian range.

METHODS

I reviewed and surveyed existing information in historical documents, technical publications and reports, agency files, and, in some cases, interviewed people with reliable first-hand knowledge of certain local waters. Principal sources of information have been listed elsewhere (Mayhood 1995, Mayhood et al. 1997*a*, 1998). The survey was substantially more complete in Alberta than in British Columbia. This material was supplemented with my own direct field observations of westslope cutthroats during my work in the Alberta portion of the native range from 1972 to the present. The number of data sources reviewed is too large to provide a comprehensive analysis in this paper, and only a brief overview is supplied here.

STOCK CHANGES AND PRESENT STATUS

Alberta

Native Stocks

Westslope cutthroat trout originally occupied the Bow and Oldman river drainages and accessible tributaries (i.e., below waterfalls and other impassable barriers), and may have extended downstream into the upper Milk River drainage of Alberta from the Montana headwaters (Prince et al. 1912, Behnke 1992). More specifically, in the Bow drainage westslope cutthroats originally were found from the extreme headwaters above Bow Lake in Banff National Park, downstream to the plains well below the city of Calgary, and from the headwaters to the mouths of its major tributaries: the Spray, Cascade, Kananaskis, Ghost, Elbow, and Highwood rivers, and Jumpingpound Creek (Prince et al. 1912, Mayhood 1995). In the Oldman drainage the subspecies was native from the falls in the headwaters below Cache Creek, downstream far out onto the plains, and from the headwaters to the mouths of the major Oldman tributaries: the Livingstone, Crowsnest, Castle, and Belly rivers, and Willow Creek. Cutthroats were excluded from almost all high mountain lakes, most of which were formed in hanging valleys and cirques by waterfalls and steep torrential reaches on the glacially steepened sides of the main valleys. Accordingly, the westslope cutthroat as a native trout in Alberta was primarily a fish of mainstem rivers and their accessible tributaries, which were mostly at low elevations. There are numerous historical records indicating that these trout were abundant throughout most of the native range in Alberta.

Present Status of Native Stocks

Westslope cutthroat now occupy considerably less than 5% of the native range in the Bow drainage, being restricted to the extreme headwaters of a few of the major tributaries and the upper mainstem (Mayhood 1995). More specifically, westslope cutthroat trout are now absent from the Bow mainstem below Lake Louise; from all but the headwaters of the Spray River; from all but the extreme headwaters of the Cascade River; from the Kananaskis River mainstem and all but 3 of its small tributaries; from all but the upper reaches and small upper tributaries of the Ghost River; from the Elbow River and all but the upper parts of 5 tributaries above the Forest Reserve boundary (1 population is known to be in decline; J. Stelfox, Alberta Fish and Wildlife, pers. comm.); from the Highwood River below the Forest Reserve boundary and from all but short, highly isolated reaches of a few tributaries; and from Jumpingpound Creek below the Forest Reserve boundary. Most remaining Bow drainage populations within the native range are tiny and isolated.

In the Oldman River drainage westslope cutthroats still occupy most of the native range in the upper Oldman basin, but have been lost from native waters in the mainstem east of the mountain front and most of its fish-accessible tributaries (Radford 1977*a*, Fitch 1977–80, Mayhood et al. 1997*a*). Most remaining populations within the native range in the upper Oldman, Livingstone, and Castle river basins appear to be reasonably large. Cutthroat trout are uncommon to rare in the St. Mary and Belly drainages (including the Waterton drainage), and have been extirpated from their native waters in the Crowsnest drainage (Fitch 1977–80, Mayhood et al. 1997*a*).

Several dozen stream populations in the Bow and Oldman systems have been examined; all but 2 show evidence of introgressive hybridization with introduced black-spotted

trouts (based mainly on examination of colouration and spotting patterns in colour photographs collected during routine fish surveys; Radford 1977*a*, Fitch 1977–80, McAllister et al. 1981, Mayhood unpubl. data). Several lake populations tested by appropriate biochemical techniques have been shown to be genetically pure (McAllister et al. 1981, Carl and Stelfox 1989), but there is reason to believe that most of these represent stocks transplanted to the lakes from local stream populations (Mayhood 1995), some of which have since been extirpated. Even if so, founder and selective effects are likely, and the present populations may no longer accurately represent the original local native stream stocks.

Introduced Stocks

Westslope cutthroats have been widely transplanted outside the native range within Alberta, including the Torrens River (Peace drainage; Nelson and Paetz 1992), some Athabasca drainage lakes and headwater streams (Ward 1974, Nelson and Paetz 1992, Barton et al. 1993), and the North Saskatchewan and Red Deer drainages (Nelson and Paetz 1992). Both genetically pure stocks and those of uncertain origin have been introduced, sometimes with other species, into formerly fishless habitat above barrier falls in several streams in the Oldman and Bow river systems. Although transplanted stocks are widespread, most individual populations appear to be small and localized. An exception is that of the Ram River (North Saskatchewan) drainage, where westslope cutthroats are widespread and abundant.

The Alberta government has transplanted westslope cutthroat trout into about 36 small, formerly fishless, high mountain lakes in the North Saskatchewan, Red Deer, Bow, and Oldman river basins. The ultimate source of nearly all of these stocks was a single population in the Spray Lakes, which no longer exists as a result of flooding from a hydroelectric reservoir constructed in the early 1950s. This stock was first introduced into Marvel Lake (Rawson 1947) and thence into Job Lake (Carl and Stelfox 1989). The latter is now the source of all westslope cutthroat trout used in Alberta's stocking programs.

British Columbia

Native Stocks

In British Columbia, the known contiguous native range of westslope cutthroat trout is the upper Kootenay River (Behnke 1992). The subspecies may also have been native in the Kicking Horse River drainage above Wapta Falls (Columbia basin), gaining access there by headwater transfer from the upper Kootenay over a low, swampy divide (M. Pole, Parks Canada, 1991, pers. comm.; Mayhood 1995), and in a few small headwater tributaries of the Columbia near Windermere and Columbia lakes (Prince 1912). There were also several small, disjunct, native populations in the South Thompson (Fraser River basin), Columbia mainstem (near Revelstoke), and Kootenay Lake basins in south-central British Columbia, mostly above barrier falls (Dymond 1932, Behnke 1992). Historical records indicate that trout, presumably of this species, were abundant throughout the contiguous native range, as were at least some of the disjunct populations (e.g., Lees and Clutterbuck 1888, Dymond 1932).

Present Status of Native Stocks

Most of the presumptive native range in the Kicking Horse drainage above Wapta Falls in Yoho National Park now lacks cutthroats (M. Pole, Parks Canada, 1991, pers. comm., Mayhood 1995). Cutthroats (mainly hybrids with introduced black-spotted trouts) still occupy all native range within the Kootenay drainage of Kootenay National Park (Alger and Donald 1984), and within the White River watershed, a large upper Kootenay tributary (Leary et al. 1987). Nevertheless, several genetically-pure westslope cutthroat trout populations are known to exist in small headwater streams within the native range in the upper Kootenay drainage in British Columbia (Leary et al. 1987). The status of most other British Columbia stocks remains undocumented, including that of most disjunct populations.

Introduced Stocks

Westslope cutthroats have been transplanted outside the native range within British Columbia, including the Pine and Narraway watersheds (Peace drainage; Nelson and Paetz 1992, Department of Fisheries and Oceans n.d.). Stocks of uncertain origin have been introduced, sometimes with other species, into formerly fishless habitat above barrier falls in several streams in the Kootenay and Columbia systems (Mayhood 1995). I presently do not have information on the status of these stocks.

THREATS

Habitat Loss/Damage

Habitat damage and loss in the Alberta range of westslope cutthroat trout began concurrently with extensive European settlement, made possible by the arrival of the Canadian Pacific Railway in the mid-1880s. By 1890 there were reports of fish-kills from sawdust pollution in the Bow River (McIllrie and White-Fraser 1983), and by 1910, concerns were voiced about the effect of dams and irrigation diversions on native trout stocks (Prince et al. 1912).

Many dams now exist within the Alberta native range of westslope cutthroat trout. These are known or suspected to be both directly and indirectly implicated in major losses of native cutthroat stocks from the Kananaskis, Spray, Cascade, Elbow, Ghost, Bow River mainstem, Belly (including the Waterton), and St. Mary rivers. A new dam on the Oldman River probably precludes restoration of native stocks within its zone of influence (Mayhood 1991). Within the British

Columbia native range, westslope cutthroats now form a negligible part of the angling catch in Lake Kookanusa (Parnell 1997), the hydroelectric reservoir formed on the Kootenay mainstem by the Libby dam in Montana. There are other smaller dams within the British Columbia native range, but I do not know their effects on westslope cutthroats.

Other major habitat problems faced by westslope cutthroat trout within the native range in Canada include: habitat damage attributable to forestry operations (British Columbia and Alberta); intensive oil and gas exploration and development (Alberta); grazing (especially in Alberta); heavy off-road vehicle use (local in British Columbia and Alberta); local urbanization (especially in Calgary, the Banff-Canmore corridor, and the Crowsnest Pass); coal mining (Crowsnest Pass and Elk River valley); and a dense network of roads throughout the range (except in the most remote parts of Banff, Yoho, Kootenay, and Waterton Lakes national parks). Cumulative impact assessments of 98 watersheds of fourth order or higher on Alberta's Eastern Slopes in the upper Oldman, Crowsnest, and Carbondale (Castle drainage) basins found that more than two-thirds are at moderate risk, and all but 3 of the remainder are at high risk, of damage from the combined effects of increased peak flows and surface erosion (sediment loading) caused by extensive clearcutting and road development (Mayhood et al. 1997*b*, Sawyer and Mayhood 1998). The extent of the existing habitat damage and the threat of continued habitat damage are both strongly affected by negligible enforcement of existing protective legislation (see below). The structure-based habitat restoration attempted so far has been generally unsuccessful, except as a short-term measure (Pattenden et al. 1998). Approaches employing improvements in land-use practices show more promise, but have not been used extensively so far (Adams and Fitch 1995).

Overfishing

Overfishing in the Alberta range of westslope cutthroat trout, like habitat damage, began concurrently with extensive European settlement, made possible by the arrival of the Canadian Pacific Railway in the mid-1880s. Obvious overfishing by means of wasteful fishing methods encompassed almost every conceivable means of capture, including netting, trapping, liming, shooting, blasting, and angling (Whitcher 1887, McIllrie and White-Fraser 1983 [regarding 1890], Prince et al. 1912). Wagonloads of trout were reportedly removed from the Castle drainage every spring by some individuals as recently as the 1920s and 1930s. It is very likely that such heavy exploitation rates contributed to rapid declines in native stocks.

Generous catch limits were introduced, then were gradually reduced over the years, but remained high for many decades in Alberta (e.g., Radford 1977*b*). These have been replaced more recently by extensive catch-and-release or large minimum-size limits (which often have an effect identical to catch and release) throughout the present westslope cutthroat range in both Alberta and British Columbia (Alberta Environmental Protection 1998, B.C. Ministry of Environment, Lands and Parks 1997). Current exploitation rates, therefore, are not likely to be a serious threat to strong remaining westslope cutthroat stocks. There are, however, many tiny remnant populations in isolated tributaries that might be adversely affected even by the low exploitation rates permitted by the new regulations.

Disease/Predation/Competition

Whirling disease is perhaps the most serious potential disease problem for wild stocks. The disease has not yet been found in Alberta (Alberta Environmental Protection 1998:20), but remains a threat to salmonid stocks throughout North America. I have no information on disease problems in British Columbia.

Introduced species of salmonids, primarily rainbow, brook (*Salvelinus fontinalis*), and (in Alberta only) brown trout (*Salmo trutta*), presently occupy most of the native range of westslope cutthroat trout where the cutthroats are now absent (Mayhood 1995, Mayhood et al. 1997*a*). The mechanism of replacement is debatable, but could include elements of differential susceptibility to angling and habitat change, competition, predation, and (in the case of rainbow trout) hybridization. This replacement process is ongoing and can be observed today, for example, in Alberta Fish and Wildlife fish survey data for the Ghost River drainage (1978 vs. 1995–96, replacement by brook trout; R. Sloan, Mount Royal College, pers. comm.) and for Quirk Creek (1978 vs. 1997–98, replacement by brook trout; J. Stelfox, Alberta Fish and Wildlife, pers. comm.).

Inadequate Protective Legislation

Canada lacks federal legislation to protect endangered species. Previously proposed legislation lacked any provisions to protect habitat of endangered species except on federal lands (mainly national parks and native reserves), which incorporate only a small part of the westslope cutthroat trout range in Canada. Future promised legislation is unlikely to improve much on this situation owing to strong resistance from the provinces of Alberta and British Columbia, among others. Alberta also lacks endangered species legislation, and is generally hostile to the concept. British Columbia can protect endangered vertebrates under its Wildlife Act and Fish Protection Act; however, its own Ministry of Environment, Lands and Parks concluded that the British Columbia government would find it difficult to argue against an "F" grade for its actual performance (Jones 1997).

The federal Fisheries Act does not provide for special protection for endangered fishes, but its habitat protection provisions, if enforced, might be strong enough to be used to

protect the habitat of fish species considered to be at risk. Unfortunately, Canada does not enforce these provisions of the Act, especially in inland waters such as those frequented by westslope cutthroat trout. The most recent parliamentary report on the government's enforcement of these parts of the Act show that the Department of Fisheries' Central and Northern Region, which includes the most severely-damaged and at-risk habitat within the westslope cutthroat's native Canadian range, has launched only 2 prosecutions in the past 3 years, and none at all in the past 2 years (Department of Fisheries and Oceans 1998). There is a complaint now before the Council on Environmental Cooperation (CEC) set up in connection with the North American Free Trade Agreement, alleging that Canada refuses to enforce the Fisheries Act. The complainants, represented by Sierra Legal Defence Fund and its U.S. equivalent, document in meticulous detail 39 separate instances of nonenforcement of clear, unquestionable, serious violations of the habitat provisions of the Fisheries Act in British Columbia alone. The CEC has decided in favour of the complainants, and is now preparing a review of Fisheries Act nonenforcement in Canada (Council on Environmental Cooperation 1998).

Hybridization/Translocation

Several nonnative black-spotted trouts (genus *Oncorhynchus*) have been widely and repeatedly introduced over many years throughout the native range of westslope cutthroat trout in Canada (Scott and Crossman 1973, Ward 1974, Nelson and Paetz 1992). Rainbow trout were especially widely stocked, but Yellowstone cutthroats were often used in the national parks, and golden trout have been stocked in a few isolated headwater lakes in Yoho and Banff parks and along Alberta's Eastern Slopes. Few biochemical genetic studies have been conducted, and those few have concentrated on assessing the genetic identity of stocks suspected to be pure (McAllister et al. 1981; Leary et al. 1985, 1987; Carl and Stelfox 1989). As noted under "Present Status" above, obvious hybrids are almost ubiquitous among the remaining stream stocks in the Oldman and Bow river basins in Alberta, and are widespread in the Kootenay watershed in British Columbia. Introduced rainbows continue to be common in the lower Bow, lower Oldman, and their lower tributaries, and are widespread in the Kootenay and Kicking Horse drainages. Yellowstone cutthroats still exist within the westslope cutthroat range in the national parks. For many years, hybrids of rainbow and local cutthroat trout were deliberately produced at the Cranbrook hatchery (Dymond 1932, Carl et al. 1959). In nearly all of the above drainages, westslope cutthroat stocks are accessible to the introduced fish. Introgressive hybridization thus continues to be a serious threat to the genetic integrity of the remaining stocks of westslope cutthroats in Canada.

The single stock used for transplanting in Alberta, the Spray–Marvel–Job Lake stock, is genetically homogeneous (McAllister et al. 1981, Carl and Stelfox 1989). The source of most westslope cutthroat trout for stocking in British Columbia is the Connor Lake population, a pure-strain but genetically uniform stock (McAllister et al. 1981, Leary et al. 1987). In many instances these stocks have been transplanted into drainages upstream of possible remnant native westslope cutthroat stocks, where they pose an introgression threat to those stocks.

PROVISIONAL CONSERVATION STATUS

Native westslope cutthroats were once widespread and abundant in the Bow, Oldman, Kootenay, and possibly the Kicking Horse drainages in Canada. Disjunct populations were native in several other small drainages of south-central British Columbia, mostly above waterfall barriers. Presently in Alberta the subspecies exists mainly as highly fragmented, isolated headwater populations (especially in the Bow basin), many of these derived from a single introduced, genetically uniform stock. Strong, but apparently hybridized, populations are common in the upper Oldman and Castle basins. Westslope cutthroats are rare in, or extirpated from, the lower Bow and many major tributaries, the lower Oldman, the St. Mary, Belly-Waterton, and Crowsnest basins. Introgressive hybridization appears to be widespread, if not almost ubiquitous among Alberta populations within the native range. In British Columbia, genetically pure native stream populations are known in a few small headwater streams of the Kootenay drainage, but apparent and confirmed introgressively hybridized stocks are widespread. The subspecies is now rare in the Kicking Horse watershed. The status of the native disjunct populations in British Columbia is not known. Canadian stocks continue to be under serious pressure from ongoing habitat damage, competitive/predative replacement from introduced species, inadequate legislation, lack of enforcement of what protective legislation does exist, and introgressive hybridization from introduced black-spotted trouts. Unless these factors are reversed, westslope cutthroat trout will continue to trend toward extinction in Canada, and should be classified as a threatened subspecies in this country. In particular, the only known genetically pure native stocks in their native habitats are rare, isolated, and subject to all of the threats cited above. These populations are in imminent danger of extinction, and should be classified as endangered in Canada. These provisional status designations should be re-evaluated when extensive, reliable population and biochemical genetic surveys have been completed throughout the Canadian range of the subspecies.

ACKNOWLEDGEMENTS

I thank R. Ament, A. Chirico, T. Clayton, D. DePape, D. Donald, L. Fitch, G. Gillund, S. Herman, T. Hurd, B. Lajeunesse, W. Marthaller, C. Pacas, J. Paczkowski, A. Paul, M. Pole, D. Radford, J. Rennels, M. Sawyer, R. Sloan, J. Stelfox, R. Walker, B. Westover, the Westwind Flyfishers, and innumerable field staff unknown to me, all of whom contributed data to this study, much of it from unpublished agency files. The work was funded in part by Parks Canada contracts KJP-01290 and KBP-2059, Morrison Petroleums Ltd., the Yellowstone to Yukon Conservation Initiative, the Alberta Sport, Recreation, Parks and Wildlife Foundation, and contributions of Alberta anglers through their licence fees. This paper is dedicated to the late G. Gillund, who made conserving Alberta's last remaining native cutthroats his final work.

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Habitat Associations of Juvenile Cutthroat Trout: Implications for Forestry Impacts

Jordan S. Rosenfeld

British Columbia Ministry of Fisheries, Research Branch
2204 Main Mall, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada
Jordan.Rosenfeld@gems4.gov.bc.ca

Marc S. Porter

British Columbia Ministry of Fisheries, Research Branch
2204 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

Eric A. Parkinson

British Columbia Ministry of Fisheries, Research Branch
2204 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

Juveniles of sea-run coastal cutthroat trout (*Salmo clarki*) rear in small coastal streams for 2–3 years before migrating to the ocean to complete their life cycle. Associations of juveniles with different habitats (e.g., pools vs. riffles) and the effect of large woody debris (LWD) on channel structure were evaluated in order to assess the long-term impact of different riparian management practices on cutthroat populations. Cutthroat densities were highest in smaller streams (<4 m bankfull width), and smaller parr (0+ year-class) were typically associated with shallower hydraulic units, while parr >110 mm long (larger 1+ and 2+ year-classes) were usually found in pools. LWD was responsible for >50% of pool formation in most streams, including those classified as S4 (<1.5 m bankfull width) under the British Columbia Forest Practices Code. Pools formed by LWD scour were, on average, 10% deeper than pools formed by other mechanisms. This indicates that long-term depletion of LWD by removal of riparian forest has the potential to seriously reduce the capacity of smaller streams to support juvenile cutthroat.

Key words: channel structure, channel width, coho, cutthroat trout, large woody debris, *Oncorhynchus kisutch*, *Salmo clarki*.

Quantifying the habitat associations of a species is an essential step in defining ecological requirements and managing wild populations. Distribution and abundance of stream fish is closely linked to channel structure, since attributes of channel structure provide critical habitat for different species or life stages (Schlosser 1987). Structure in streams arises from both substrate heterogeneity and the natural tendency of running water to form pools, riffles, and meanders (Leopold et al. 1964), and is enhanced by the presence of obstructions such as large woody debris (LWD) and boulders, which cause flow convergence and scour (Lisle 1986).

Juveniles of anadromous coastal cutthroat trout (*Salmo clarki*), along with coho salmon (*Oncorhynchus kisutch*), rear in first to fourth order streams (Hartman and Gill 1967, Chamberlin et al. 1991), where channel structure, particularly that associated with LWD, plays an essential role in salmonid life history (Hartman et al. 1996). Scour created by LWD is a major pool-forming mechanism, and LWD may also provide an important refuge from predation. Side channels, backwaters, pools, and structure associated with LWD and rootwads provide critical overwintering and rearing habitat, by providing hydraulic refuges during winter floods (Bustard and Narver 1975*a*,*b*) and pool habitat during summer low flow.

Channel structure in small coastal streams is often significantly altered by both natural and anthropogenic processes. Land-use practices such as logging, agriculture, and urbanization can have pervasive effects on channel structure, ranging from increased sedimentation to bank instability and alteration of LWD input rates from the riparian zone. Smaller streams in particular may be severely impacted by logging because a greater proportion of the watershed may be cut at 1 time (Jones and Grant 1996), and because riparian regulations provide less protection for very small streams (e.g., the British Columbia Forest Practices Code requires no riparian buffers for streams <1.5 m bankfull width; Province of British Columbia 1995). Similarly, urban development is typically concentrated in valley bottoms, where small coastal streams are most abundant.

The overall project goal is to quantify cutthroat habitat relationships and develop predictive functions that can be

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

used to model the effects of channel structure modifications on cutthroat distribution and abundance. Our initial objectives were to define the freshwater habitat associations of juvenile cutthroat both within and between streams, and to assess the role of LWD in the creation of cutthroat habitat.

METHODS

Study Sites

Forty-five sites on 44 streams with populations of anadromous coastal cutthroat trout were sampled at summer base flow during June–September 1997 and 1998. Coho parr were also present at 31 of the sites. Twenty-two stream sites were located on the Sechelt Peninsula of the lower mainland of British Columbia, 40 km NW of the city of Vancouver, within a 30-km radius of the town of Sechelt. The remaining 23 stream sites were located on the west coast of Vancouver Island, within a 30-km radius of the village of Tofino.

In order to represent the range of habitats used by anadromous cutthroat trout, sites were chosen to encompass a broad range of channel sizes (1.2–17.0 m bankfull width), logging histories (clearcut, second growth, and old growth), basin gradients (high or low), and channel complexity.

Data Collection

Fish were collected by triple-pass electroshocking of stop-netted sections using a backpack electrofisher, with voltage set between 300 and 500 volts to maximize capture efficiency while minimizing injury to fish. Shocked areas were left undisturbed for at least 45 minutes between passes, with the same shocking time (±10%) and operator during each pass. Stop-netted reaches were chosen to be representative of the average channel characteristics of the stream reach, and included 3–14 discrete hydraulic (channel) units. In order to assess associations of fish with discrete channel unit types, individual (contiguous) channel units were isolated using multiple stop-nets at 23 of the 45 sites sampled. All other sites were stop-netted at the upper and lower ends of the sample reach. Fish were anaesthetized using carbonated water, measured to the nearest mm, and weighted to the nearest 0.1 g after each pass, and released following completion of fish collection.

Canopy closure was visually estimated at each site; average site gradient was measured using a Sunnto clinometer; stream temperature was measured to the nearest degree C using an alcohol thermometer; and conductivity was measured to the nearest µSiemen with a hand-held meter (Cole-Parmer TDStestr-30). Sites were classified as either clearcut (logged within 15 yr), second growth (logged 15–100 yr ago), or old growth (unlogged or with minimal pre-industrial selective harvest), based on the age of the riparian forest and air photographs. Average channel structure was characterized by measuring habitat features over a reach length of 20 bankfull channel widths that included the electroshocked reach. Channel units were classified as pools (0% gradient, low current velocity, relatively deep), glides (0–1% gradient, flow, minimal water surface turbulence), runs (higher current velocity, turbulent flow), riffles (1–3% gradient, higher current velocity, water surface broken by protruding substrata, relatively shallow), or cascades (>3% gradient, higher current velocity, water surface broken by larger substrate particles), as described in Johnston and Slaney (1996) and Moore et al. (1997). Maximum depth, minimum depth (lower riffle crest if the channel unit was a pool), average width, and substrate composition (visually estimated in 5 size classes [fines: <2 mm; gravel: 2–64 mm; cobble: 64–250 mm; boulder: 250–4,000 mm; and bedrock: >4,000 mm]) were measured in each channel unit. If the channel unit was a pool, the pool-forming mechanism was also recorded as either boulder scour, LWD scour, bank scour, or free-form (Montgomery et al. 1995).

Data Analysis

Cutthroat and coho population size at each site was estimated from the removal data using the Schnute maximum likelihood procedure adapted to a 3-pass depletion (T. Johnston, B.C. Ministry of Fisheries, pers. comm.; Schnute 1983). Time constraints prevented a third pass at 2 of the sites, where populations were estimated using a Zippin removal estimate (Zippin 1958). Low fish numbers precluded depletion estimates of fish abundance for individual channel units. Numbers of fish collected in all 3 passes within a stop-netted channel unit were instead summed and divided by channel unit area to give minimum density estimates within individual channel units.

To assess the determinants of total cutthroat abundance at a site, cutthroat density was modelled (SAS General Linear Model procedure; SAS 1989) as a linear function of site (bankfull width, percent pool, channel type, dominant substrate class, b-axis of the largest particle moved by the channel; Province of British Columbia 1996) and drainage basin characteristics (high, medium, or low gradient) for the subset of 33 sites where all of these habitat factors were measured.

To define cutthroat habitat associations within a channel, juveniles were divided into 3 size classes for analysis (small: <80 mm; medium: 80–110 mm; and large: >110 mm). These groupings represent distinct classes in the size-frequency distribution of cutthroat collected from streams where individual channel units were stop-netted, and roughly correspond to 0+, 1+, and 1+ to 2+ or greater year-classes of fish. Selective use of different channel units as a function of maximum channel unit depth was evaluated by comparing observed frequency distributions for each size class with the

Table 1. Results of multiple regression of total cutthroat abundance versus habitat variables measured at sample sites (R^2 = 0.62, p = 0.001).

Source	Degrees of freedom (df)	Sum of squares (SS)	Mean sum of squares (MS)	*F* ratio	Probability (*P*)
bankfull width[a]	1	1.71	1.71	7.0	0.014 *
substrate	3	2.26	0.75	3.1	0.046 *
percent pool[b]	1	1.65	1.65	6.8	0.015 *
b-axis	1	1.62	1.62	6.7	0.017 *
basin gradient	2	2.19	1.10	4.5	0.020 *
error	24	5.83	0.24		
TOTAL	32				

[a] log-transformed
[b] arcsin (square root) - transformed

frequency distribution of maximum channel unit depth available in the environment (n = 188 individually stop-netted channel units). Significant deviation in maximum channel unit depth of occupied channel units relative to available channel units was evaluated using a chi-squared test.

Average density per channel unit type per site was calculated for the same subset of streams where individual channel units were stop-netted (n = 23). Preference for different habitats in terms of differences in densities of fish between channel unit types (for the 3 size classes of cutthroat) were evaluated using a 1-way ANOVA (n = 23) for the 3 different size classes. Only pool (n = 23), glide (n = 17), run (n = 5), and riffle (n = 20) channel unit types were included in the analysis, since cascades occurred too infrequently for meaningful analysis.

RESULTS

Total densities (per wetted area) of both cutthroat and coho juveniles were highest in smaller stream channels (Fig. 1). Maximum densities occurred in channels <4–5 m bankfull width.

Total cutthroat density was significantly related to channel bankfull width, dominant substrate size, percent pool in the electroshocked reach, b-axis of the largest particle moved at high flow, and basin gradient class (Table 1). These factors collectively accounted for 62% of the total variance in cutthroat abundance at 33 sites (P = 0.001).

Frequency of use of channel units of different maximum depth (deeper channel units tended to be pools, shallow ones were riffles, and runs and glides were intermediate in depth) was size-class dependent. Smaller cutthroat (<80 mm) used channel units of varying maximum depths roughly in proportion to their background availability (Fig. 2). Intermediate-sized cutthroat (80–100 mm) showed a disproportionately greater use of deeper channel units (Fig. 2; χ^2 = 28.2, P = 0.00), and larger cutthroat (>110 mm) showed strong selection for the deeper channel units (χ^2 = 27.9, P = 0.00).

Minimum density estimates of small cutthroat were not significantly different between channel unit types (Fig. 3), although densities were lowest in pools. Densities of intermediate-sized cutthroat trout were highest in pools and lowest in riffles (P = 0.04), and densities of larger cutthroat were also higher in pools than in riffles (P = 0.001). The apparent preference of cutthroat for pools was clearly size-related,

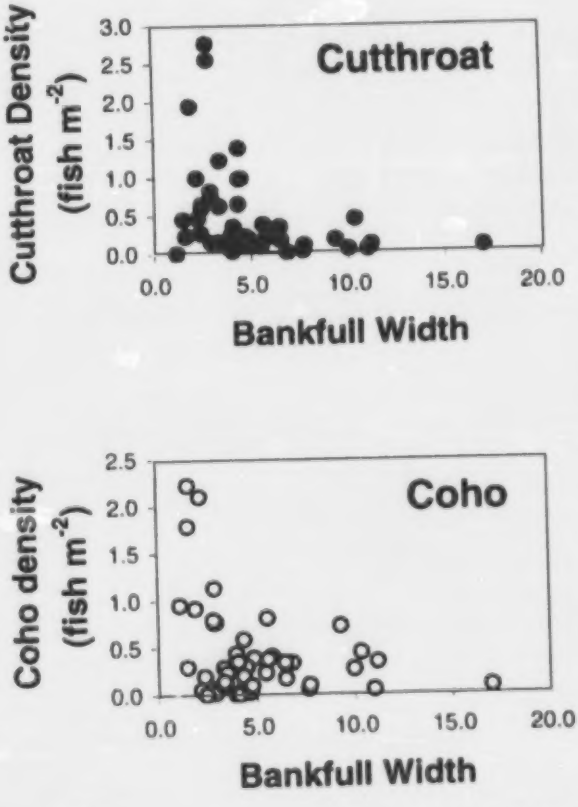

Figure 1. Density (per wetted stream area) of cutthroat trout and coho salmon juveniles as a function of bankfull channel width.

with densities of small, medium, and large size classes being, respectively, 0.7, 2.6, and 6.4 times greater in pools than in riffles. Coho also occurred in pools at densities 3 times greater than in riffles (Fig. 3).

The proportion of pools formed by scour associated with LWD was usually >50% across a wide range of channel widths (1.2–11.0 m; Fig. 4). Maximum channel depth was, on average, 10% deeper ($P = 0.005$) in pools that were formed by LWD scour (mean depth 38.8 cm) than in pools that were formed by other mechanisms (free-form meander, bank or boulder scour; mean depth 35.0 cm).

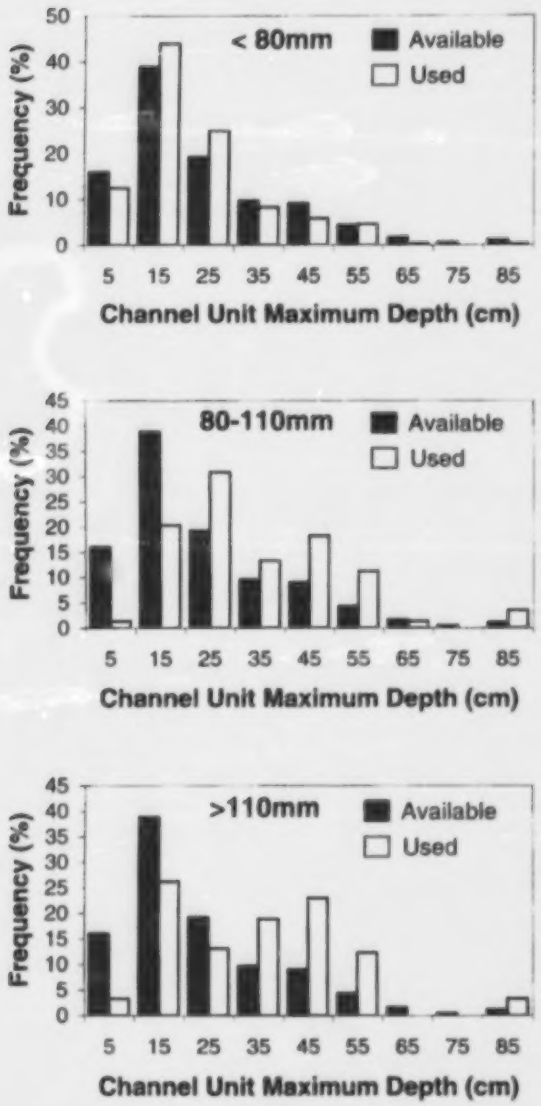

Figure 2. Frequency of occurrence of small (<80 mm; $n = 686$), medium (80–110 mm; $n = 143$), and large (>110 mm; $n = 122$ mm) cutthroat trout in channel units ($n = 188$) of different maximum depth.

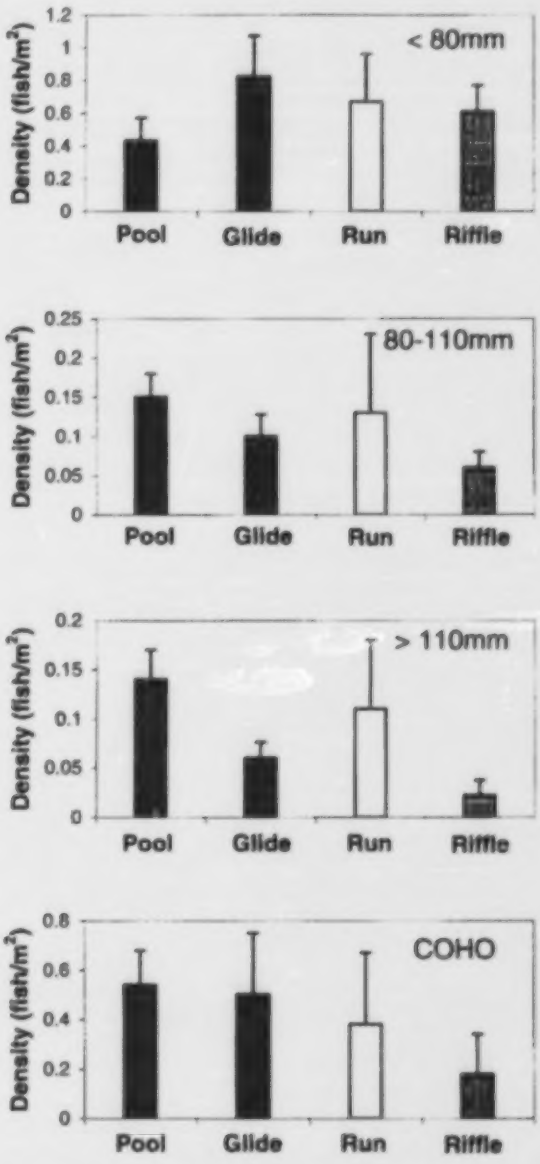

Figure 3. Density of small (<80 mm; $n = 686$), medium (80–110 mm; $n = 143$), and large (>110 mm; $n = 122$ mm) cutthroat trout and coho salmon juveniles in pools ($n = 23$ streams), glides ($n = 17$), runs ($n = 5$), and riffles ($n = 20$).

DISCUSSION

Juvenile cutthroat (and coho) were most abundant in smaller streams (<4 m bankfull width), which is consistent with previous observations of cutthroat ecology and distribution. Hartman and Gill (1967) found cutthroat to occupy smaller, low-gradient drainages in the lower Fraser valley, and De Leeuw and Stuart (1981) similarly observed highest densities of anadromous cutthroat in very small coastal streams (<3 m channel width).

Multiple regression analysis (Table 1) also indicates that cutthroat densities are significantly higher in lower gradient drainage basins with pool-riffle channel structure and gravel substrate. These streams typically occur in low gradient, alluvial, valley-bottom topography, areas that unfortunately are subject to the greatest impacts from logging, urbanization, and agriculture (Healey and Richardson 1996, Healey 1997). This pattern of human settlement has resulted in extensive degradation or loss of fish habitat, and the designation of anadromous cutthroat trout as a species with many stocks potentially at risk (Slaney et al. 1996).

The greater frequency of use (Fig. 2) and higher densities (Fig. 3) of larger-sized cutthroat (and coho) in deeper channel units is also consistent with previous observations of cutthroat habitat preference (Glova 1986). Larger cutthroat parr may avoid shallower channel units because of a higher vulnerability to predation (Harvey and Stewart 1991), higher energetic costs associated with holding in a swifter current (Facey and Grossman 1992), or because of a restricted visual field when feeding on drift in shallower turbulent water. Deeper channel units may provide greater cover associated with water depth or LWD, and have lower energetic costs, while providing a larger visual field for detecting prey items in the drift or on the water surface. Smaller cutthroat may be able to find suitable drift-feeding territories in shallower riffle or glide habitat, because of their smaller territory size (Grant and Noakes 1987) associated with a lower absolute food requirement relative to larger fish. Alternatively, the apparent lack of selection for pool habitat by smaller cutthroat may be driven by a high risk of predation in pools, since larger cutthroat will prey on smaller cutthroat as well as on coho parr.

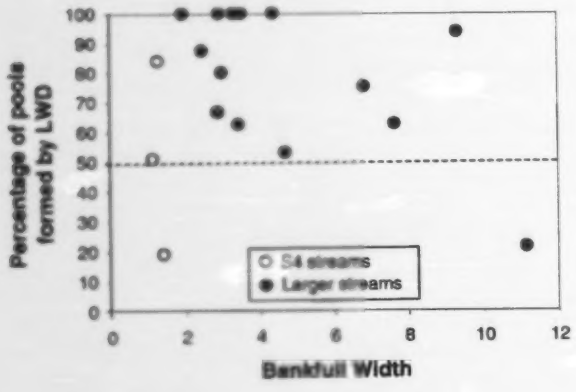

Figure 4. Percentage of pools formed by large woody debris as a function of bankfull channel width. S4 streams have a bankfull channel width of <1.5 m.

Because cutthroat parr rear for 2–5 years in coastal streams (Scott and Crossman 1973), the availability of deeper pool habitat for 1+ or 2+ year-class fish is the most likely habitat factor limiting freshwater productivity. This assumes that the streams preferred by cutthroat are not limited for spawning or young-of-the-year rearing habitat, which appears to be a reasonable assumption for lower-gradient (1–5%) gravel-cobble channels. Higher densities of coho in pools also indicates that the abundance of pool habitat may often be the most significant physical habitat factor limiting coho capacity as well.

There is a large amount of literature documenting the pervasive influence of LWD on channel structure (e.g., Bisson et al. 1987, Bilby and Ward 1989, Richmond and Fausch 1995), in particular the role of LWD in pool formation. Observations from our survey indicate that most of the pools in the sampled streams were formed by scour associated with LWD (Fig. 4). Although this does not demonstrate that pool frequency would be lower in the absence of LWD, it is consistent with other studies that have demonstrated an increase in pool frequency with higher LWD loading (Montgomery et al. 1995). Pools formed by LWD scour are also, on average, deeper than pools formed by other mechanisms, indicating that pools formed by LWD are of higher "quality"—assuming that greater depth and the presence of wood are reliable correlates of habitat quality to fish (e.g., Harvey and Stewart 1991). Small increases in pool depth may be very important for population persistence during episodic droughts with a long return period, when small coastal streams run dry and the entire freshwater population may be restricted to 1 or 2 deeper pools.

Although increased nutrients and decreased light limitation following logging may result in short-term increases in primary production, invertebrate production, and fish biomass (Murphy and Hall 1981, Murphy 1995), production usually declines to pre-disturbance levels following canopy closure (usually 10–15 years; Murphy and Meehan 1991). In many instances production will decline well below pre-disturbance levels if there has been significant degradation of channel structure, in particular excessive sedimentation (Hartman and Scrivener 1990) or loss of pool habitat.

Given that pools are likely to limit juvenile cutthroat (and coho) abundance in many small coastal streams, land management practices that reduce LWD input rates will lead to long-term reduction in habitat capacity as LWD in the

channel decomposes without replacement (Murphy and Koski 1989). Channel structure in many coastal streams in British Columbia may continue to degrade over the next century, until clearcut riparian forests regrow and begin recruiting to the stream channel. In order to maintain undegraded channel structure, forested landscapes need to be managed so that LWD input rates approximate natural background inputs. Full riparian buffers will maintain LWD input rates from riparian (streamside) sources, which may be the major source of LWD to smaller, low-gradient streams. Other major sources of LWD include debris torrents from hillslopes (Hogan et al. 1998), and, potentially, S5 and S6 streams (fishless streams which require no riparian reserve zone under the B.C. Forest Practices Code) in coastal zones, where these tributary streams have the power to transport LWD downstream to fish-bearing reaches. Protecting and maintaining upslope sources of LWD may be equally or more important than maintaining riparian sources, depending on the relative importance of these different LWD sources for a given stream channel.

Our observations also indicate that LWD may be an equally important pool-forming mechanism in permanently wetted S4 streams (fish-bearing channels <1.5 m bankfull width), although our data set only includes 3 S4 streams (Fig. 4). Although the current British Columbia Forest Practices Code requires limited retention of riparian forest along S4 streams, and suggests retention of all conifer stems <30 cm DBH (diameter at breast height) in streams dependent on LWD to maintain channel processes, it is largely discretionary (Province of British Columbia 1995). Permanently wetted S4 streams (as opposed to smaller, seasonally dry ones) often constitute valuable rearing habitat for juvenile cutthroat trout and coho. Given that permanently wetted S4 streams are likely to provide as good habitat as S3 (1.5–5 m bankfull width) streams, and are likely to be partly dependent on LWD for pool formation, a largely intact riparian zone may be necessary to ensure long-term recruitment of LWD and maintenance of channel structure.

ACKNOWLEDGEMENTS

We would like to thanks R. Ptolemy for advice during the course of this study, and D. Bates and the Sechelt Indian Band for assistance selecting and sampling sites. M. Roberge, T. Lee, D. Forsyth, and B. Matsuda provided valuable assistance in the field. This project was funded by Forest Renewal British Columbia.

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Declines in the Abundance of Thompson River Coho Salmon in the Interior of Southern British Columbia, and Canada's Coho Recovery Plan

James R. Irvine
Fisheries and Oceans Canada, Pacific Biological Station
3190 Hammond Bay Road, Nanaimo, BC, V9R 5K6, Canada
irvinej@pac.dfo-mpo.gc.ca

Michael Bradford
Fisheries and Oceans Canada
c/o School of Resource and Environmental Management
Simon Fraser University, Burnaby, BC, V5A 1S6, Canada

ABSTRACT

Populations of unenhanced coho salmon (*Oncorhynchus kisutch*) in the North and South Thompson River watersheds have declined at rates >50% per generation since 1988. In recent years, <5 female spawners/km were seen on average in these streams. Of a sample of 34 extant populations in 1988, no coho were observed in 11 of these in 1997. Populations were judged to be at a high risk of biological extinction. To conserve these populations, managers were advised that fishing mortality should be as close to zero as possible. In 1998, major changes were made to salmon fisheries in British Columbia. No directed fisheries on wild stocks of coho were permitted, and there was mandatory non-retention of coho. In areas where Thompson coho were expected to be prevalent, only a few experimental salmon fisheries were allowed. In areas where Thompson coho were not expected to be prevalent, the only salmon fisheries permitted were directed at the other species. These restrictions appear to have improved spawning escapement in some areas. Conservation must continue to be a high priority in the management of Pacific salmon to reduce the risk of extinction for populations of Thompson coho salmon.

Key words: coho salmon, conservation, extinction, Fraser River, *Oncorhynchus kisutch*, risk, Thompson River.

Coho salmon (*Oncorhynchus kisutch*) are widely distributed throughout the North Pacific basin (Sandercock 1991). Coho is an economically important species, contributing to commercial, recreational, and aboriginal catches along the Pacific coast of North America. Unfortunately, numbers of coho salmon are declining in many regions, and some populations have become extinct (e.g., Nehlsen et al. 1991, Weitkamp et al. 1995, Slaney et al. 1996, Northcote and Atagi 1997).

In the United States, the Endangered Species Act considers a species to be endangered if it is in danger of extinction throughout all or a significant portion of its range (Weitkamp et al. 1995). A species is threatened if it is likely to become endangered within the foreseeable future. The U.S. National Marine Fisheries Service proposed 6 evolutionarily significant units (ESUs) for coho salmon extending from central California to southern British Columbia. Coho were considered to be in danger of extinction in 1 ESU (central California coast), likely to become so in 4 ESUs, and not presently in danger of extinction, nor likely to become so, in 1 ESU. Small et al. (1998*a*,*b*) recently examined genetic variation among coho salmon populations in British Columbia. They found coho salmon from the upper Fraser and Thompson River watersheds in south-central British Columbia to be genetically distinct from other British Columbia coho, and proposed that coho salmon in the upper Fraser and Thompson River drainages form an ESU.

In this paper we examine trends in abundance of unenhanced coho salmon populations in the Thompson River, a major tributary to the Fraser River, arguably the largest salmon producer in the world. We also describe fishery management changes made by Canada's Department of Fisheries and Oceans (DFO) during 1998 to conserve threatened stocks of coho salmon.

STUDY AREA AND METHODS

The Thompson River watershed is large and diverse. At Kamloops, the North Thompson from the north and the South Thompson from the east join to form the Thompson River. We assembled a time series of spawner escapement estimates for >50 streams from the North and South Thompson watersheds. Most of these estimates used data

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Table 1. Annual (r_{an}) and finite rates of change (per year = $1\text{-}e^{r_{an}}$; per generation = $1\text{-}e^{3r_{an}}$) for coho stock aggregates from the North and South Thompson watersheds.

	Annual			Finite					
Stock aggregate	75-88 Annual	88-97 Annual	75-97 Annual	75-88 Per Year	88-97 Per Year	75-97 Per Year	75-88 Per Gen.	88-97 Per Gen.	75-97 Per Gen.
South Thompson -19 streams	0.09	-0.26	-0.04	0.09	-0.23	-0.40	0.30	-0.54	-0.11
North Thompson - 8 streams	0.04	-0.27	-0.06	0.04	-0.23	0.06	0.13	-0.55	-0.18

gathered by DFO officers and other DFO staff, and were based on multiple stream walks. Although the accuracy of most estimates was unknown, we felt many were relatively precise, and when aggregated by geographic area, provided valuable trends on spawner abundance. The data that we considered reliable extended to 1975.

We filtered our escapement database to remove confounding effects of inconsistent monitoring. We used data only from unenhanced systems for which we had 18 annual escapement estimates (out of a maximum of 23). To account for missing estimate(s), we used the average of the numbers preceding and following the missing data. If a missing datum was at the beginning or end of a time series, we used the estimate for the nearest year. The result was a time series of spawner abundance estimates for 19 streams from the South Thompson drainage, and 8 streams from the North Thompson.

We converted our estimates of spawner escapements to numbers of females/km of stream accessible to migrating salmon. The DFO is attempting to develop limit reference points for coho using these units. We assumed the proportion of females among spawners to be 0.45 based on data in Irvine et al. (1999).

To examine trends in escapement, the slope of the regression of the natural log of escapement on year r_{an} was used to estimate the annual intrinsic rate of change of each stock aggregate. Finite rates of change per year were calculated as $1\text{-}e^{r_{an}}$ and per generation rates as $1\text{-}e^{3r_{an}}$ (assuming all fish had a 3-year life cycle consisting of 1 winter in freshwater and 1.5 summers at sea). Positive numbers indicated an increasing trend while negative numbers indicated a declining trend.

This approach does not illustrate the fate of many of the small populations that comprise these aggregates. To examine this, we determined which streams had population estimates in both 1988 and 1997, and determined whether the estimates for these streams increased or decreased.

RESULTS

In the South Thompson, female spawner densities were moderate but variable from 1975 through 1983, and generally at higher levels until about 1989, after which they declined (except 1992) (Fig. 1). Escapements to the unenhanced North Thompson aggregate (Fig. 2) followed a similar temporal pattern. Modest returns from the mid-1970s to 1983 preceded about 7 years of generally higher densities. Since 1991, spawner numbers and densities have been relatively low, with the exception of 1992. For each aggregate, spawner numbers in 1997 exceeded those in 1996; 1997 densities were less than the parental brood escapement (i.e., 1994) for South Thompson streams, and almost identical to the brood year for North Thompson streams. Spawner densities were lower in the South Thompson than they were in the North Thompson.

The rates of change for the South and North Thompson, respectively, were +30 and +13% per generation during 1975–1988 and -54 and -55% per generation since 1988 (Table 1).

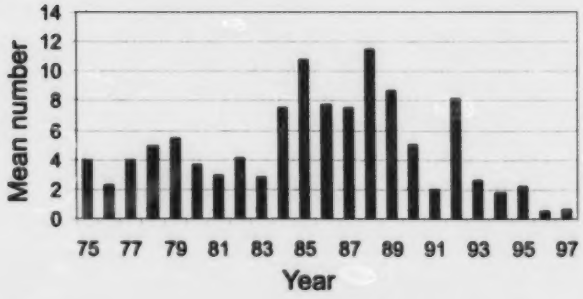

Figure 1. Mean numbers of female coho/km accessible habitat in 19 streams in the South Thompson watershed.

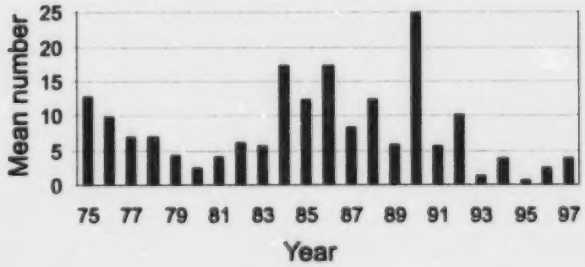

Figure 2. Mean numbers of female coho/km accessible habitat in 8 streams in the North Thompson watershed.

The fate of individual populations during the period that large generational declines occurred (i.e., 1988–1997) can be seen by plotting results for those streams with fish count data for 1988 and 1997 (Fig. 3). A few populations increased during this period, but the vast majority decreased, and about 30% of the streams with coho extant in 1988 had no coho observed in them during 1997. For more detailed results, see Bradford (1998) and Irvine et al. (1999).

DISCUSSION

Numbers of coho salmon returning to spawn in streams in the Thompson River watershed declined significantly, particularly during the most recent decade. Returns the last 2 years to unenhanced streams in the South Thompson ranged between 15 and 25% of longer term averages, while returns to North Thompson streams were only about one-third of previous returns.

If populations continue to decline, local although perhaps temporary extinctions in individual streams are inevitable. In 1997, no spawning coho were seen in about 30% of streams where coho had been seen in 1988. Note that spawning coho can be in a stream, but not observed. Surveyors typically only examine portions of a stream, adult coho tend not to occur in high densities, and their cryptic colouration makes coho difficult to see. Nevertheless, coho are clearly in danger of extinction in some streams in the Thompson River.

Straying will reduce extinction rates. However, coho appear to have relatively little genetic exchange among populations. For instance, Labelle (1992), studying 14 coho populations on Vancouver Island, found that most strays contributed <1% of the escapement. Sandercock (1991) summarizes several studies that document that coho usually exhibit very low rates of straying from home streams. Small et al. (1998 *a,b*), using microsatellite DNA analysis, found a correlation between rates of gene flow within the Thompson/Fraser and the geographic proximity of coho populations. Coho salmon from the North, South, and lower Thompson regions could be differentiated by allele frequency.

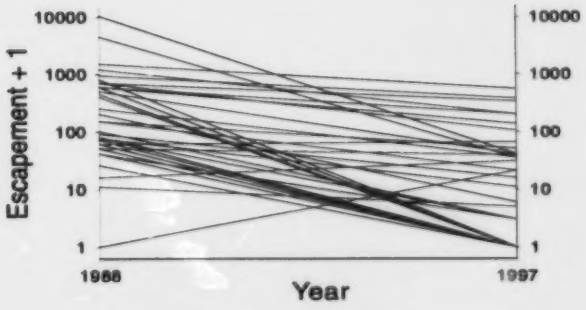

Figure 3. Fate of 34 spawning populations from the North and South Thompson watersheds between 1988 and 1997 showing the variation in the rate of decline among populations. Many streams have declined to "none-observed" status for this broodline in 3 generations.

We are unsure of the likelihood or frequency of streams in the Thompson River being recolonized from nearby populations. To minimize social, genetic, and economic costs associated with the loss of coho populations in individual streams, and to maximize potential rebuilding rates, we (Bradford 1998, Irvine et al. 1999) recommended maintaining as large a population base within the watershed as possible. This recommendation meant that fishing mortality should be as close to zero as practicable.

Low and declining marine survivals (e.g., Coronado and Hilborn 1998), combined with overfishing in mixed stock fisheries, have contributed to the decline of Thompson coho (Bradford 1998, Irvine et al. 1999). Because coho spend approximately 1 year in freshwater before migrating to the ocean, they are also sensitive to changes in their habitat. There are many examples of habitat degradation causing reductions in carrying capacity for coho in the Thompson watershed (e.g., Harding et al. 1994, Department of Fisheries and Oceans 1997). To best conserve these fish, we need to better understand the role of habitat alterations.

MANAGEMENT IMPLICATIONS

The status of Thompson coho was reviewed (Irvine et al. 1999), and a risk assessment undertaken (Bradford 1998) early in 1998. The Pacific Stock Assessment Review Committee (PSARC) accepted our findings and advised that Thompson River coho were extremely depressed and would continue to decline even without fishing mortality under current marine survival conditions, and that some populations were at high risk of biological extinction (Stocker and Peacock 1998). On 21 May 1998, David Anderson, Minister of Fisheries and Oceans Canada announced that, "Despite significant conservation measures implemented by my department over the last 3 years, scientific evidence demonstrates conclusively that wild coho stocks are declining and some are at extreme risk." Minister Anderson proclaimed a conservation objective of zero fishing mortality for critical Thompson (and upper Skeena) coho stocks. On 19 June 1998, he and Pierre S. Pettigrew, Minister of Human Resources Development Canada, identified significant new management measures for the 1998 salmon fishery. Federal funding of $400 million was provided to help rebuild the resource, restructure the fishery, and help people and communities adjust to the changing fishery.

Regulatory changes made to salmon fisheries in 1998 to

conserve threatened coho stocks were probably the most significant ever implemented within the Pacific Region of Canada. In 1998, no directed fisheries on wild stocks of coho were permitted, and mandatory non-retention of coho was implemented in all areas. The coast of British Columbia was divided into a series of Red and Yellow zones. In the south, Red zones consisted of those areas and times where Thompson coho stocks were expected to be prevalent. Yellow zones were areas where Thompson coho were not expected to be prevalent. In Red zones, only a few restricted experimental fisheries for commercial, aboriginal, and recreational sectors were allowed; these fisheries were closely monitored. In Yellow zones, the only salmon fisheries permitted were directed on salmon species other than coho. Many fisheries involved modifications to gear types, and all were carefully monitored to ensure that coho bycatches were minimal.

Preliminary information suggests that fishing restrictions improved coho spawning escapements in some areas. Restrictive fishery management measures to conserve threatened coho stocks are needed for at least the next 5–7 years. On 9 January 1999, Minister Anderson announced that conservation would continue to be the priority in the management of Pacific salmon. Ongoing diligence is required to minimize the risk of extinction for Thompson River and other endangered populations of coho salmon.

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Predictive Models of Fish Species Distribution in the Blackwater Drainage, British Columbia

Marc S. Porter

British Columbia Ministry of Fisheries, Research and Development Section
2204 Main Mall, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada
Marc.Porter@gems1.gov.bc.ca

Jordan Rosenfeld

British Columbia Ministry of Fisheries
Research and Development Section
2204 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

Eric A. Parkinson

British Columbia Ministry of Fisheries
Research and Development Section
2204 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

Management of fish biodiversity requires the ability to understand and predict expected species distributions. Models predicting species distributions can give insight into habitat requirements and expected probabilities of encountering species in unsampled areas, and help identify unique outlier populations and potential biodiversity hot spots. Previous research has shown that large-scale geomorphic variables can be linked to fish habitat use and used as predictors of fish species occurrence. Our goal was to determine whether reliable models of species distributions could be developed for freshwater fish in British Columbia, using large-scale macrohabitat data linkable to GIS (geographic information system) map coverages. We surveyed 48 streams in a representative watershed with high species diversity (the Blackwater) and developed statistical models based on macrohabitat variables to predict the distribution of 15 fish species found within the drainage. Correct classification rates of our logistic regression models based solely on map-based variables were generally high for most species found in the Blackwater (ranging from 73% to 90%) and showed only marginal improvement with inclusion of field-based information. The models correctly predicted the presence of individual species at stream sites 82.2% of the time (on average) when applied to an independent data set from the Blackwater drainage, but were much less successful when applied to data from a geographically distant watershed (the Similkameen). Further refinements to quantify interregional variation in habitat conditions, however, should allow the development of regional fish distribution models as a GIS layer in British Columbia's new watershed atlas.

Key words: Blackwater, distribution model, fish, geographic information system, logistic regression, macrohabitat, Similkameen.

Conserving fish biodiversity requires reliable information on the existing distribution and habitat use of individual fish species. This is especially true for species that are at risk because they exist at low densities, are endemic or disjunctly distributed, or are sensitive to the cumulative effects of agriculture, urbanization, or forestry (McPhail and Carveth 1992, Cannings and Ptolemy 1998). Although distributions and habitat requirements have been fairly well researched for most game species in British Columbia, those of many nongame species are poorly documented. A focused effort to systematically define fish species distributions and habitat use is therefore critical if management of fish and fish habitat are to be effectively integrated into land-use planning (Parkinson et al. 1996).

The identification of habitat features that characterize fish distributions and abundance has long been a primary objective of fisheries research, but has been largely focused on salmonids (e.g., Bjornn and Reiser 1991, Meehan and Bjornn 1991). Additionally, much of this work has attempted to characterize habitats through traditional small-scale approaches, such as stream-reach or habitat-unit inventories (e.g., Fausch and Northcote 1992, Nickelson et al. 1992). It may be more important, for planning purposes, to determine fish distributions and response to habitat using a spatially larger-scale geomorphic approach (Nelson et al. 1992, Rieman and McIntyre 1995, Kruse et al. 1997). Previous research has shown that

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large-scale geomorphic variables (e.g., elevation, channel slope, stream size) can be linked to fish habitat use and used as reliable predictors of fish species occurrences (Gorman and Karr 1978, Lanka et al. 1987, Nelson et al. 1992, Rieman and McIntyre 1995, Kruse et al. 1997, Watson and Hillman 1997). The development of incidence functions, based on such large-scale variables, could be an important tool for fisheries and forestry managers, permitting prediction of the presence or absence of particular fish species, even where little detailed habitat or population information is available (Rieman and McIntyre 1995, Kruse et al. 1997). Models predicting species distribution can also give insight into large-scale habitat requirements and help identify unique outlier populations and potential biodiversity hot spots. Although predictions of presence-absence will need to be verified with ground surveys, predictive models can help focus inventory and management activities on areas where species are considered likely to occur and may allow managers to anticipate the possible consequences of large-scale habitat change.

The purpose of this study was to quantitatively evaluate macrohabitat use by individual fish species in the Blackwater River drainage, a representative high diversity watershed in central British Columbia, and to identify the physical variables that are important in discriminating between stream sites with and without individual fish species. Our primary objective was to move beyond the use of expert opinion to predict the presence of different fish species at a site, and to develop predictive habitat models that are suitable for interfacing with GIS (geographic information system)–based planning and information systems. We developed logistic regression models using a combination of map-based features (e.g., watershed gradient, drainage area) and data collected in the field (e.g., bankfull width, stream temperature) to determine the level of information necessary to reliably predict fish species distributions. We then tested the derived relationships using independently collected data from the Blackwater drainage itself and from a distant watershed (the Similkameen River drainage), to determine the utility of such models for management purposes.

STUDY AREA

The Blackwater River is located on the Fraser Plateau of British Columbia's central Interior (Fig. 1) and drains an area of approximately 12,400 km² (Water Survey of Canada 1989). Climate in the area is characterized as rather dry with cold winters and short summers (Griffith 1991). The existing forest in the drainage is dominated by lodgepole pine (*Pinus contorta*), white spruce (*Picea glauca*), and black spruce (*Picea mariana*), and includes Interior Douglas-fir (*Pseudotsuga menziesii*) in the wetter areas. Varied degrees of timber management (including extensive clearcut logging) and limited agriculture in valley bottoms occur throughout much of the Blackwater drainage. At least 19 species of fish are known to occur within the Blackwater, making it an ideal watershed to assess habitat use by multiple species.

METHODS

Data Collection

Sampling took place during summer low-flow periods (July–September) at 36 stream sites in 1996 and at an

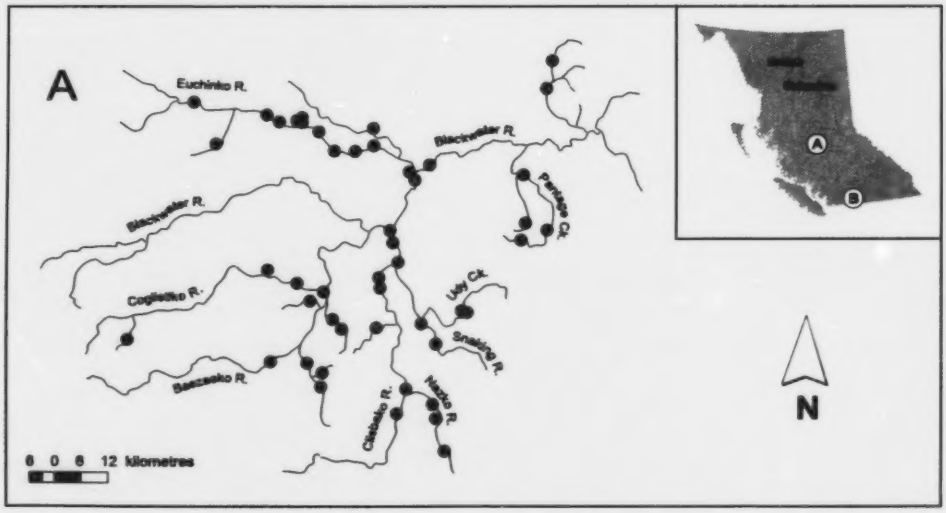

Figure 1. Location of the Blackwater River drainage (A) in north-central British Columbia. Also indicated is the location of the Similkameen River drainage (B) which was used as a testing area for Blackwater-derived fish models.

Table 1. Physical habitat variables used to characterize sites in the Blackwater drainage. Information for map-based variables was derived solely from 1:50,000 topographic maps; field-based variables required measurements at a site by a ground-based field crew.

Variable	Units	Type	Description
Drainage area	km^2	Map based	Total area (± 1 km^2) from which water drains into the stream, upstream of the survey site.
Watershed gradient	%	Map based	Drop in elevation over length of stream from headwaters to site of sampling (± 0.5%).
Reach gradient	%	Map based	Drop in elevation over length of reach encompassing survey site (defined by 100 ft. contour change bracketing site) (± 0.5%).
Elevation	m	Map based	Site elevation (± 15 m) as determined from topographic maps and/or hand held GPS units.
Distance from a lake > 4ha	km	Map based	Linear distance from site (± 0.1 km) to nearest large lake (following along the natural meanders of the stream).
Distance from the Fraser R.	km	Map based	Linear distance from site (± 0.1 km) to the nearest major river (Fraser R.) (following along the natural meanders of the stream).
Lake area in drainage	ha	Map based	Total surface area (± 0.1 ha) of all lakes present within the defined drainage area.
Sinuosity		Map based	Channel meander length over downvalley linear distance.
Site gradient	%	Field based	Drop in water surface elevation over total length of stream channel surveyed (± 0.5%).
Bankfull width	m	Field based	Average width (± 0.1 m) of the bankfull channel (defined by the presence of permanent rooted vegetation).
Bankfull depth	cm	Field based	Average depth (± 1 cm) from the height of the bankfull channel to the channel bottom.
Maximum wetted depth	cm	Field based	Deepest point in stream channel (± 1 cm).
Maximum temperature	°C	Field based	Highest temperature recorded at site (± 0.1°C) by hourly temperature logger readings.
Degree-days	°C*days	Field based	Area under curve of days monitored versus mean daily temperature for a site recorded over a single year.

additional 12 sites in 1997. We surveyed a mixture of small, intermediate, and large streams to cover the full range of flowing water habitats in the drainage. Sampling sites were of variable size but included all naturally occurring habitat units (e.g., riffles, pools, runs, glides, cascades, etc.) present within a stream reach (defined in Johnston and Slaney 1996). Fish were collected at all sites using a Smith-Root backpack electroshocker. At deeper sites, 10-m-long sinking gillnets with 7.5-cm, 5.0-cm, 2.5-cm, and multi-panel (1.25–3.25-cm) mesh sizes were set parallel to the current. Gill nets were generally left in place for 2–4 hours and were checked regularly during the set to limit fish mortality. Minnow traps, seine nets (both pole and beach), trap nets, and angling were used as additional sampling techniques where appropriate. All or some of these sampling methods were employed to ensure a high rate of capture success for all species present at a particular site. Captured fish were identified to species in the field, measured to the nearest mm total length, weighed to the nearest 0.1 g, and released. Unidentifiable fish were measured, preserved in 10% formalin, and later identified in the laboratory. Species not captured after extensive sampling with the full range of applicable sampling techniques were assumed to be absent from a site for the purposes of data analysis.

Physical characteristics of each site were measured based on procedures outlined in Newbury and Gaboury (1993) and Johnston and Slaney (1996). Fourteen site habitat variables (Table 1) were calculated using information derived either from 1:50,000 topographic maps or by direct measurement in the field. The site habitat variables selected (e.g., bankfull width, drainage area, gradient measures) were those previously identified as useful indicators of fish species occurrence (Lanka et al. 1987, Rieman and McIntyre 1995, Kruse et al. 1997, Watson and Hillman 1997), many of which are also correlated with habitat features at a finer scale (e.g., current velocity, substrate type, channel stability, pool-riffle frequencies) that are known to influence fish distribution and abundance (Gorman and Karr 1978, Angermeier and

Karr 1983, Moyle and Baltz 1985, Lobb and Orth 1991, Hubert and Kozel 1993).

DATA ANALYSIS

Statistical models that could predict presence-absence of individual species were developed using binary logistic regression. Logistic regression was used because of the binomial nature of the dependent variable (fish presence-absence) and because the analysis provides a probabilistic prediction of fish presence. The analysis is insensitive to differences in fish densities at sites, and was biased only if a fish species used a stream site but was never detected. We believed we had minimized this possibility by sampling each site extremely thoroughly, and by using a suite of different sampling techniques for both juvenile and adult fish. Logistic regression analysis, unlike most other multivariate procedures, does not require that variables be normally distributed, linearly related, or of equal variance within each group (Press and Wilson 1978, Tabachnick and Fidell 1996). With 2 possible outcomes (in this case, fish species presence or absence) the form of the logistic function is:

$$P = e^u/(1 + e^u),$$

where P = the estimated probability, e = the inverse natural logarithm of 1, and u = linear model:

$$u = A + b_1X_1 + b_2X_2 + \ldots + b_nX_m,$$

where A = regression constant, b_n = regression coefficients, and X_m = independent variables.

Before generating logistic regression models, Pearson correlation matrices were calculated to test for multicollinearity among the site variables. Multicollinearity occurs when variables are too highly correlated, which can result in both logical and statistical problems in a multivariate analysis (Tabachnik and Fidell 1996). As a number of the variables were highly correlated with each other, we selected a subset of variables for use in logistic regression analysis that: 1) were relatively uncorrelated ($| r | < 0.7$); and 2) had the highest correlation with excluded variables (Eadie and Keast 1982). Four map-based variables (drainage area, watershed gradient, reach gradient, and elevation) and 2 field-based variables (bankfull width and maximum temperature) were identified as independent habitat dimensions using this screening process.

To illustrate univariate differences between sites with and without individual fish species, mean values for the selected habitat variables were compared with parametric t-tests, using log-transformed ($\log_e[x+1]$) data to meet assumptions of normality and homogeneity of variance. Logistic regression models were subsequently fit for both untransformed and log-transformed variables; both sets of variables produced similar results, but the untransformed data produced better fitting models and are the only results presented here. SYSTAT LOGIT (Steinberg and Colla 1991) was used for all logistic regression analyses.

Statistical significance of the models was assessed using a chi-square test ($P < 0.05$) and the goodness-of-fit of each model was evaluated by the magnitude of McFadden's Rho^2 (Steinberg and Colla 1991). Rho^2 is a summary statistic indicating the accuracy with which the model fits the observed data, analogous to the R^2 in a linear regression model. Rho^2 values are always between 0 and 1, and a higher Rho^2 corresponds to more significant results. Due to the binary nature of the dependent variable in logistic regressions, however, correlations between predictors and the dependent variable are much lower than would be obtained in a linear regression. Consequently Rho^2 values obtained may seem quite low, yet still represent a good model fit. Rho^2 values between 0.20 and 0.40, for example, are considered very satisfactory (Steinberg and Colla 1991). Models were also assessed using the percent correct-classification (the proportion of sites correctly classified by the model), a more readily interpretable criteria. Probabilities for species occurrence were calculated from the logistic models and compared with observed values. Predicted probabilities of ≥0.50 indicated species presence, whereas probabilities of <0.50 indicated species absence. A high correspondence between predicted and observed presence-absence indicated a good model fit. Correct-classification rates assigned in this manner are somewhat biased, as the classified sites were also used to estimate model parameters. Recommended cross-validation of the classification rates (Diefenbach and Owen 1989) using random subsets of the data was not possible, however, due to the small number of sites used in the analysis.

Two sets of logistic regression models were constructed for each species, 1 using solely map-based variables (i.e., drainage area, watershed gradient, reach gradient, and elevation), and a second incorporating the map-based variables and additional habitat variables collected in the field (i.e., maximum stream temperature and/or bankfull width). Our objective was to assess the relative predictive ability of models based on map-based versus combined map-and-field-based variables. A number of different map-and-field-based models were produced using combinations of the different field-collected variables, and model quality was judged based on relative Rho^2 values.

To test model performance in predicting fish species occurrences, logistic regressions generated from our surveys were applied to 2 independent data sets. The first data set was derived from previous fish sampling of 31 sites within the Blackwater drainage (Griffith 1991), and was used to assess the reliability of the models within their own region. As sampling methodologies used in this earlier survey were not intensive enough to reliably conclude fish species absence at a sampling site, the information was used only to test model performance in predicting fish presence. Five species of fish were sampled at enough sites to allow an assessment of the

species models. Predicted probabilities of ≥0.5 indicated probable fish presence, whereas probabilities of <0.5 indicated absence. We tested only the map-based logistic models (using habitat information derived from 1:50,000 topographic maps) for the sampled fish species, as only limited habitat information was available from the earlier survey. The second data set was derived from fish and habitat surveys undertaken at 46 sites in a geographically distant watershed (the Similkameen River drainage; Rosenfeld 1996), and was used to test the broad-scale applicability of the specific regressions generated for the Blackwater. The Similkameen is a high-diversity watershed located in southwestern British Columbia (Fig. 1), which shares at least 7 fish species in common with the Blackwater drainage (i.e., prickly sculpin [*Cottus asper*], largescale sucker [*Catastomus macrocheilus*], longnose dace [*Rhinichthys cataractae*], mountain whitefish [*Prosopium williamsoni*], northern squawfish [*Ptycheilus oregonensis*], rainbow trout [*Oncorhynchus mykiss*], and redside shiner [*Richardsonius balteatus*]). Intensive sampling was undertaken at all sites in the Similkameen and was considered sufficient to be reflective of both fish species presence and absence. Additionally, habitat information from these surveys allowed us to test both map-based and map-and-field-based logistic models. Finally, to test the reliability of individual habitat predictors across regions we generated unique logistic regressions for the Similkameen data set using the same habitat variables employed in the Blackwater models. The quality of model fit for these new regressions was assessed by correct classification rates for each sampled fish species.

RESULTS

Habitat models were constructed for 14 species from a total of 5,190 fish collected at 48 stream sites in the Blackwater drainage in 1996 and 1997. Three additional species (lake chub [*Cousesius plumbeus*], Pacific lamprey [*Entosphenus tridentatus*], and bull trout [*Salvelinus confluentus*]) were collected from too few sites (<5) to perform meaningful analyses. At least 1 of the measured habitat variables was significantly different between sites with and without fish for all species (Table 2), with the exception of longnose sucker (*Catastomus catostomus*). Significant logistic regression models based on the 4 independent map-based site habitat variables (drainage area, watershed gradient, reach gradient, and elevation) were produced for each of the fish species surveyed in the Blackwater, again with the exception of longnose sucker (Table 3). The overall correct classification rates of map-based models ranged from a minimum of 73% for burbot (*Lota lota*) and chinook (*Oncorhynchus tshawytscha*), to 90% for leopard dace (*Rhinichthys falcatus*), northern squawfish, and rainbow trout. Overall correct classification rates averaged 82.0% (±6.0% SD) for the 13 fish species that generated significant regressions, and the average Rho^2 value for these models was 0.387 (±0.132 SD).

Eleven out of 13 map-based models were better at correctly classifying sites without an individual species than sites with the species. This difference in correctly classifying species absent versus present sites ranged from as little as 1% for northern squawfish to as much as 33% for chinook. Models for only 2 species (mountain whitefish and rainbow trout) showed better success at classifying sites with the species rather than those without them. The discrepancy in these cases ranged from 10% for mountain whitefish to 55% for rainbow trout.

The addition of field-based variables in the logistic models generated improvements in Rho^2 values for 10 of the 14 species, ranging from 0.029 (bridgelip sucker [*Catostomus columbianus*]) to 0.425 (prickly sculpin). The best map-and-field-based models showed an average improvement of 0.126 (±0.111 SD) in Rho^2 values versus the corresponding map-based models for these 10 species. Addition of bankfull width to the map-based variables generated the best logistic model for bridgelip sucker, chiselmouth (*Acrocheilus alutaceus*), leopard dace, mountain whitefish, and redside shiner. Inclusion of both maximum temperature and bankfull width variables generated the best regressions for chinook, large-scale sucker, and peamouth chub (*Mylocheilus caurinus*). Addition of maximum stream temperature alone generated the best regression for prickly sculpin. Logistic regression models for burbot, longnose dace, and rainbow trout showed no improvement with the incorporation of independent field variables. Inclusion of field variables failed to generate a significant regression for longnose sucker.

Model improvement following inclusion of field variables was not as apparent based on overall classification success (Table 3). Although classification rates did improve considerably for prickly sculpin (+15%) and redside shiner (+7%) with inclusion of the best predictor field variables, the other species showed only a minimal improvement (0–4%), or else displayed even worse classification success (i.e., bridgelip sucker = -2%, leopard dace = -5%). The average improvement in classification rates across the 10 species was only 1.8% (±5.2% SD) relative to the models using only map-based variables.

Five species (i.e., rainbow trout, chinook salmon, longnose dace, burbot, and northern squawfish) sampled at 31 sites in an earlier survey of the Blackwater drainage (Griffith 1991) were used as independent data to test map-based logistic regression models. Correct classifications of fish presence from the independent data ranged from 62.5% for burbot to 100% for longnose dace and northern squawfish (Table 4). The average percentage agreement between actual and predicted fish species presence for the 5 species was 82.2% (±17.7% SD).

Seven species were common to both the Blackwater and Similkameen drainages, permitting testing of the generality of our logistic models at a larger scale. Overall correct

Table 2. Macrohabitat variables for stream sites with individual fish species present (P) or absent (A). Variable means for each species were compared using parametric t-tests after $\log_e(x+1)$ data transformation. Symbols next to variables indicate significance level (NS = not significant; * = $p < 0.05$).

Species	N	Habitat Variables					
		Drainage area (km^2)	Watershed Gradient (%)	Reach Gradient (%)	Elevation (m)	Bankfull Width (m)	Maximum Temperature (°C)
Bridgelip Sucker (P)	16	3039.5 *	1.0 *	0.7 *	870.4 *	31.7 *	20.7 *
Bridgelip Sucker (A)	32	592.9 *	2.0 *	1.3 *	936.8 *	11.9 *	19.4 *
Burbot (P)	16	2382.6 *	0.9 *	0.8 (NS)	875.9 (NS)	24.1 *	20.5 (NS)
Burbot (A)	32	916.3 *	2.1 *	1.3 (NS)	935.4 (NS)	16.0 *	19.7 (NS)
Chinook (P)	17	2492.4 *	1.2 (NS)	1.2 (NS)	884.7 (NS)	20.2 (NS)	19.6 (NS)
Chinook (A)	31	789.4 *	1.9 (NS)	1.0 (NS)	929.6 (NS)	18.4 (NS)	20.5 (NS)
Chiselmouth (P)	12	3634.8 *	0.9 *	0.6 *	836.0 *	39.5 *	21.2 *
Chiselmouth (A)	36	562.8 *	1.9 *	1.3 *	943.1 *	10.4 *	19.3 *
Large-scale Sucker (P)	13	3520.3 *	0.9 *	0.6 *	839.3 *	41.3 *	21.4 *
Large-scale Sucker (A)	35	696.9 *	1.9 *	1.3 *	941.7 *	10.6 *	19.3 *
Leopard Dace (P)	12	3610.1 *	1.0 *	0.5 *	837.1 *	38.3 *	21.2 *
Leopard Dace (A)	36	573.2 *	1.9 *	1.4 *	942.6 *	11.0 *	19.3 *
Longnose Dace (P)	17	2662.8 *	1.0 *	1.0 (NS)	887.9 (NS)	26.0 *	20.5 (NS)
Longnose Dace (A)	31	600.1 *	2.1 *	1.2 (NS)	927.5 (NS)	14.0 *	19.5 (NS)
Longnose Sucker (P)	17	1452.1 (NS)	1.6 (NS)	1.0 (NS)	895.6 (NS)	17.3 (NS)	20.6 (NS)
Longnose Sucker (A)	31	1492.4 (NS)	1.6 (NS)	1.2 (NS)	920.8 (NS)	20.1 (NS)	19.7 (NS)
Mountain Whitefish (P)	24	2420.7 *	1.0 *	1.0 (NS)	891.5 (NS)	29.0 *	20.4 (NS)
Mountain Whitefish (A)	24	406.4 *	2.3 *	1.2 (NS)	935.9 (NS)	7.9 *	19.2 (NS)
Northern Squawfish (P)	17	3002.2 *	0.9 *	0.5 *	832.6 *	34.9 *	21.4 *
Northern Squawfish (A)	31	531.7 *	2.0 *	1.5 *	963.8 *	9.3 *	18.9 *
Peamouth Chub (P)	12	3227.0 *	1.0 *	0.6 *	837.0 *	33.9 *	21.0 *
Peamouth Chub (A)	36	809.0 *	1.9 *	1.3 *	939.7 *	13.4 *	19.5 *
Prickly Sculpin (P)	18	1491.8 *	0.9 *	0.5 *	854.4 *	35.0 *	21.2 *
Prickly Sculpin (A)	30	1471.3 *	2.0 *	1.4 *	940.6 *	11.6 *	19.2 *
Rainbow Trout (P)	40	1675.4 *	1.3 *	1.1 (NS)	910.7 (NS)	20.5 (NS)	20.1 (NS)
Rainbow Trout (A)	8	348.7 *	3.2 *	1.3 (NS)	918.8 (NS)	11.2 (NS)	19.8 (NS)
Redside Shiner (P)	20	2775.1 *	1.1 *	0.5 *	852.7 *	33.0 *	20.9 *
Redside Shiner (A)	28	430.0 *	2.0 *	1.6 *	961.9 *	7.9 *	18.9 *

classification rates for the species models were low, ranging from only 48% (mountain whitefish) to 72% (northern squawfish, large-scale sucker, and redside shiner) using the Blackwater map-based models, and ranging from 53% (mountain whitefish) to 73% (large-scale sucker and northern squawfish) using the map-and-field-based models (Table 5). The average overall correct classification for sites in the Similkameen was 65.9% (±8.5% SD) for the map-based models and 67.3% (±6.9% SD) for the map-and-field-based models.

Unique logistic regressions were generated for species in the Similkameen using the same predictor variables that were tested in the Blackwater models (stream temperature at time of summer sampling in the Similkameen replacing maximum stream temperature derived from data loggers in the Blackwater). Overall correct classification rates for these models were very high, ranging from 78% for rainbow trout to 93% for northern squawfish and prickly sculpin using map-based models, with improvements to as much as 98% for prickly sculpin and large-scale sucker when using map-and-field-based models (Table 5). The average overall correct classification for sites in the Similkameen using these unique regressions was 86.1% (±6.2% SD) for the map-based models and 88.1% (±8.2% SD) for the map-and-field based models.

DISCUSSION

Logistic regression models can be used both to identify the specific habitat features that are associated with species

Table 3. Logistic regression models predicting fish species presence-absence in the Blackwater drainage. Models are based either solely on map-derived variables or a combination of map-derived and field-measured variables. Model significance (P-value), McFadden's Rho^2, and model correct classification percentage (present/absent/overall) are given for each species. Rho^2 values between 0.2 and 0.4 are considered to represent satisfactory fits. Models incorporating field derived variables are presented only if they showed an improvement in Rho^2 versus the species map-based model.

Species	Model Type	P-value	McFadden's Rho^2	Correct Classification %		
				Present	Absent	Overall
Bridgelip Sucker	Map based	0.002	0.276	72	87	81
	Map + Field based[a]	0.002	0.305	72	83	79
Burbot	Map based	0.003	0.246	68	76	73
	Map + Field based	-	-	-	-	-
Chinook Salmon	Map based	0.013	0.196	53	86	73
	Map + Field based[c]	0.023	0.295	65	74	69
Chiselmouth	Map based	<0.001	0.475	79	88	85
	Map + Field based[a]	<0.001	0.603	79	88	85
Large-scale Sucker	Map based	<0.001	0.399	64	85	79
	Map + Field based[c]	<0.001	0.554	86	77	81
Leopard Dace	Map based	<0.001	0.486	79	94	90
	Map + Field based[a]	<0.001	0.559	71	91	85
Longnose Dace	Map based	0.006	0.224	63	83	75
	Map + Field based	-	-	-	-	-
Longnose Sucker	Map based	0.877	N/A	N/A	-	-
	Map + Field based	-	-	-	-	-
Mountain Whitefish	Map based	<0.001	0.417	92	82	88
	Map + Field based[a]	<0.001	0.545	96	86	92
Northern Squawfish	Map based	<0.001	0.677	89	90	90
	Map + Field based[a]	<0.001	0.750	95	93	94
Peamouth Chub	Map based	0.001	0.332	62	89	81
	Map + Field based[c]	0.005	0.393	85	83	83
Prickly Sculpin	Map based	<0.001	0.410	75	81	79
	Map + Field based[b]	<0.001	0.835	94	95	94
Rainbow Trout	Map based	0.002	0.425	98	43	90
	Map + Field based	-	-	-	-	-
Redside Shiner	Map based	<0.001	0.468	82	81	81
	Map + Field based[a]	<0.001	0.553	86	89	88

[a] best combined model based on inclusion of bankfull width only
[b] best combined model based on inclusion of maximum stream temperature only
[c] best combined model based on inclusion of bankfull width and maximum stream temperature

presence and to quantitatively predict the probability of occurrence of individual species. Watson and Hillman (1997) suggested that basin-scale features, such as watershed size, geology, and valley type, function in concert to provide the flow regimes and in-stream channel conditions (channel size, substrate, etc.) that any given species of fish requires for rearing, spawning, and migration. Individual drainages should, therefore, exhibit a specific and predictable array of large-scale physical characteristics that different fish species require in order to persist. Earlier application of logistic models indicates that fish distributions can be successfully explained and predicted based on these large-scale macrohabitat variables. Logistic regressions incorporating only channel slope and elevation, for example, correctly classified 87% of cutthroat trout sites in western Wyoming (Kruse et al. 1997). Watson and Hillman (1997) also produced a significant logistic model predicting presence-absence of bull trout at stream sites in Washington, Idaho, and Montana, and concluded that watershed-scale physical processes and spatial landscape patterns were the most important factors

Table 4. Agreement between actual and predicted fish species presence for the independent data set (Griffith 1991) used to test the map-based logistic regression models of fish distribution in the Blackwater drainage.

Fish species	No. of sites	No. correctly classified	Agreement (%)
Rainbow Trout	22	18	81.8
Chinook Salmon	18	12	66.7
Longnose Dace	11	11	100.0
Burbot	8	5	62.5
Northern Squawfish	5	5	100.0

determining the species' distribution throughout the western United States.

In this study we developed logistic regression models for predicting fish species distributions in the Blackwater drainage based solely on a limited set of large-scale macrohabitat variables (Table 1). High overall correct classification rates for many of our species models (ranging from 69 to 94%) indicated that, in general, we had successfully captured a set of reliable, quantitative variables for predicting the probability of occurrence of individual fish species at particular streams during summer low-flow. The ability to derive significant logistic regression models is likely to be greatest for specialist species with narrow habitat requirements. Failure to generate successful predictive regressions for certain fish species (e.g., longnose sucker) suggests that it may be more difficult to develop incidence functions for highly generalist species. This could possibly be rectified by increasing sample size to improve the model's statistical power, or by incorporation of additional variables, thereby more clearly defining the habitat niche of the species. This may require the addition of habitat variables that are not captured at a macrohabitat scale.

The predictive success of our fish species models was generally improved by inclusion of field-based variables into the regressions. However, the magnitude of improvement over strictly map-based models was small, showing an average increase of only 1.8% across all species. Although habitat conditions at a smaller scale (e.g., substrate, % pool, large woody debris abundance, etc.) may be important determinants of fish abundance (Bowlby and Roff 1986, Jowett 1990, Watson and Hillman 1997), they may be of limited importance in determining presence-absence. Our results indicate that logistic models based solely on large-scale, map-derived variables may be sufficient for purposes of predicting fish species distributions within watersheds and that little is to be gained by collecting field-based information for prediction purposes. Consistent with this contention, our map-based species models were correct (on average) 82.2% of the time when classifying stream sites that had been independently sampled for fish species presence within the Blackwater drainage in 1990.

Our attempt to apply specific species models outside the geographic boundaries of the Blackwater drainage was not successful. The average overall correct classification for stream sites sampled in the Similkameen River drainage using the Blackwater-generated regressions was low for both map-based (65.9%) and map-and-field-based (67.3%) species models. Although these results are better than would be expected by chance, they indicate that the relationship between habitat features and fish distributions varies across broad land classes. Regional differences in landscape morphology are probably too great to allow wide use of the particular regression coefficients generated from the Blackwater sites. However, unique species regressions generated for the Similkameen drainage produced much higher correct classification rates (ranging from 78 to 98%) for these same sites. This indicates that the selected macrohabitat variables have appreciable predictive power across drainages, but may require modification of the regression coefficients on a regional basis through continued testing, ground-truthing and revision of species regressions.

Table 5. Overall correct classification rates of fish species occurrence (presence-absence) at sites in the Similkameen watershed for: I) Blackwater generated logistic models; and II) unique Similkameen logistic models (using the same habitat parameters as the Blackwater regressions).

Model	n	Overall correct classification (%) of Similkameen sites						
		Prickly Sculpin	Large-scale Sucker	Longnose Dace	Mountain Whitefish	Northern Squawfish	Rainbow Trout	Redside Shiner
I) Blackwater models								
Map Based	46	65	72	67	48	72	65	72
Map + Field based	46	70	73	-	53	73	-	70
II) Similkameen models								
Map based	46	93	91	80	83	93	78	85
Map + Field based	46	98	98	-	85	93	-	85

Although we modelled fish presence as a function of distribution at summer base-flow, fish show strong seasonal variation in habitat use (Schlosser 1982, 1985; Ross et al. 1985; Angermeir and Schlosser 1989), and additional models could be developed to predict spawning or overwintering distributions. Logistic models should also be refined to include factors such as the presence of downstream barriers (Kruse et al. 1997) that will influence theoretical distributions. It might also be useful to develop age-specific regressions, as habitat use and associated distributions may vary greatly between juveniles and adults of some species (Northcote 1997). Finally, biotic interactions such as competition and predation will also influence fish distributions (Watson and Hillman 1997), and the presence and/or relative densities of other fish species could be incorporated as independent variables in logistic regression.

MANAGEMENT IMPLICATIONS

Most studies of the effects of land-use practices on fish populations have been focused at relatively small spatial scales, generally involving small to moderate-sized stream reaches. However, to understand fully the effects of land-use practices on fish communities, a broader spatial context such as the watershed basin or sub-basin scale may be required. Watson and Hillman (1997) suggested that watershed analysis nested within a hierarchical framework of physical processes and management standards is the best choice for protecting fish biodiversity and maintaining the habitat of sensitive fish species. As part of the longer term goal of integrated watershed management in British Columbia, the Blackwater and Similkameen watersheds were selected as test sites for the development of predictive fish distribution models. The statistical models generated appear to adequately explain the individual distributions of most fish species within these drainages, using only macrohabitat variables derived from large-scale maps. Although inclusion of field-based information often improved the predictive abilities of the species distribution models, improvements were generally marginal. Further refinement of regression parameters could lead to the development of predictive models, based strictly on remotely derived variables, that could map out provincial fish distributions based on probability of occurrence. Regional-scale models, based predominately on geomorphic variables and the habitat use of individual species, can be incorporated into geographic information system coverages like the *British Columbia Watershed Atlas* (Anon. 1996), allowing fisheries managers to assess and anticipate potential land-use impacts on fish populations. Such fish distribution models have the potential to provide a powerful diagnostic and predictive tool for improving watershed planning and management practices. Further development of these models might ultimately incorporate a nested hierarchy of habitat information, defining an individual species' habitat use at increasingly finer scales of spatial resolution.

ACKNOWLEDGEMENTS

This project was conducted with funding from Forest Renewal British Columbia and the Habitat Conservation Trust Fund. Our appreciation goes to M. Heise, J. Pritchard, A. Salomon, B. Wicks, P. Van Dishoek, and M. Roberge for enthusiastic and able assistance in the field. D. McPhail and G. Haas provided useful advice, and M. Lirette provided essential local support and expertise. We would also like to thank P. and C. Carter for their unrivalled hospitality.

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A Case History: The Kokanee Stocks of Okanagan Lake

Bruce G. Shepherd[1]
British Columbia Ministry of Environment, Lands and Parks
Suite 201 - 3547 Skaha Lake Road, Penticton, BC, V0H 1K0, Canada

ABSTRACT

Within a single decade, Okanagan Lake kokanee (*Oncorhynchus nerka*) went from being managed for sport fishery purposes to being a major concern from the perspective of conservation biology. Both shore-spawning and stream-spawning runs have declined dramatically, despite a fishery closure since 1995. Shore spawners are genetically different from stream spawners, and are quite unique in spawning behaviour. The 1998 returns of both stocks were about 1% of the levels of the 1970s, when routine monitoring of spawning began. While the introduction of an exotic species, the opossum shrimp (*Mysis relicta*), is likely the driving force behind the decline, there are a variety of other forces that may have initially masked and more recently aggravated the situation. A long-term (20-year) action plan for the recovery of the kokanee stocks was formulated in 1995. Phase I (1996–2000) of the plan stands on 2 legs: 1) developing a practical means of *Mysis* control; and 2) conserving key habitats. The results of the first 3 years of work on the plan are reviewed, and options for the future discussed. Given intensifying human development pressures in the Okanagan, can we save habitats that are currently empty for future fish?

Key words: kokanee, limnology, *Mysis relicta*, Okanagan, *Oncorhynchus nerka*, opossum shrimp, spawning habitat.

First Nations peoples fished both sockeye and kokanee (*Oncorhynchus nerka*) spawning runs in the Okanagan system long before white pioneers first appeared in the Okanagan Valley, B.C. The early settlers took an abundance of kokanee for granted, and with rakes and washtubs used to take kokanee home by the wagon-load from the beaches and tributaries of Okanagan Lake. Even into the late 1960s, kokanee remained so numerous that no need was seen even for an angling bag limit province-wide. By the late 1970s, concern for Okanagan Lake kokanee stocks began to build, as angler catches climbed and spawner numbers shrank. In the mid-1980s, the British Columbia Environment Ministry began to allocate additional resources to improving the management of the Okanagan Lake kokanee sport fishery, which had an estimated net worth of about $1,000,000 annually (Shepherd 1990).

By 1995, stock conservation concerns had replaced those of fishery management, and Okanagan Lake was closed to fishing for kokanee effective 15 March 1995. That closure is unlikely to be rescinded in the near future, given the bleak population monitoring results to date. Both stream and shore spawner numbers have continued to decline since annual escapement counts began in the early 1970s. In 1998, spawner numbers were down to disastrously low levels of <13,000 adults in total, just 1% of the numbers seen 25–30 years ago (Fig. 1). Annual hydroacoustic (Fig. 2) and trawling (Fig. 3) surveys of younger age classes indicate that, without some significant changes, further declines, rather than recovery, are likely in the coming years.

SELECTED FEATURES OF OKANAGAN LAKE KOKANEE STOCKS

Both stream-spawning and shore-spawning strains of kokanee occurred naturally in the Okanagan Lake system, probably due to glacial wanderings some 10,000 years ago (McPhail and Carveth 1992). Despite this relatively recent arrival in the Okanagan, the 2 strains differ markedly from each other in terms of spawning and incubation lifestyles (Table 1). The stream-spawning kokanee runs are similar to the "standard" sockeye in spawning morphology and behaviour, and can be quite variable in size and age at maturity. In contrast, shore spawners are smaller and less variable in size (Table 1), and are much less pronounced with regard to spawning colouration (Dill 1996) and other secondary sexual maturation features (e.g., dorsal hump, hooked jaws, compression of body).

Okanagan Lake shore spawners are unique among salmon in their spawning behaviour. Rather than rounded gravel, they choose large, angular rocks in shallow (15–100 cm water depth) shoreline areas, presumably to allow water circulation

[1] Present address: Habitat and Enhancement Branch, Fisheries and Oceans Canada, 417 – 2nd Avenue W, Prince Rupert, BC, V8J 1G8, Canada
shepherdb@pac.dfo-mpo.gc.ca

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Table 1. Selected characteristics of shore-spawning and stream-spawning kokanee in Okanagan Lake (modified from Shepherd and Sebastian 1998).

Characteristic	Shore spawners	Stream spawners
Spawning dates		
Peak	25 Oct–30 Oct	1 Oct–5 Oct
Range	15 Oct–4 Nov	21 Sep–20 Oct
Preferred substrates	• angular riprap • size >5 cm	• rounded gravels • size <5 cm
Spawning behaviour	• no obvious pairing • little defence of redd • school when disturbed • "bright" spent fish common	• pairing • redds defended • hide in cover • spent are "rags" and highly coloured
Nose–fork lengths (1987–97)	236 mm overall mean (216–250 mm range)	265 mm overall mean (229–352 mm range)
ATU[a] - hatch (range) - emerge	616 C-days[a] (550–682) 880 C-days	660 C-days (627–693) 902–970 C-days
Hatching date	6 Jan–15 Mar	1 Jan–30 Apr
Emergence date	15 Mar–15 Apr	1 Apr–15 Jun

[a] ATU = Accumulated Thermal Units. For this paper, the ATUs were calculated by the cumulative addition of mean daily water temperatures measured in degrees centigrade above freezing (C-days). For example: Eggs spawned on 15 Sep. Mean daily temperature on 16 Sep = 10°C, so ATU = 10 C-days. Mean daily temperature on 17 Sep = 11°C, so ATU = 10 + 11 = 21 C-days. Mean daily temperature on 18 Sep = 8.5°C, so ATU = 10 + 11 + 8.5 = 29.5 C-days. Etc.

to the eggs via wave action without shifting the substrate. Shore spawners have abandoned the standard salmon strategy of an approximate 2-week period of spawner pairing, male defence of the female against other males, and female defence of her redd until death. Instead, schools of adults move onto the shoreline in the morning, and females drop out of the schools to investigate the substrate. Once a female finds a suitable spot, she will circle and clean the selected area using

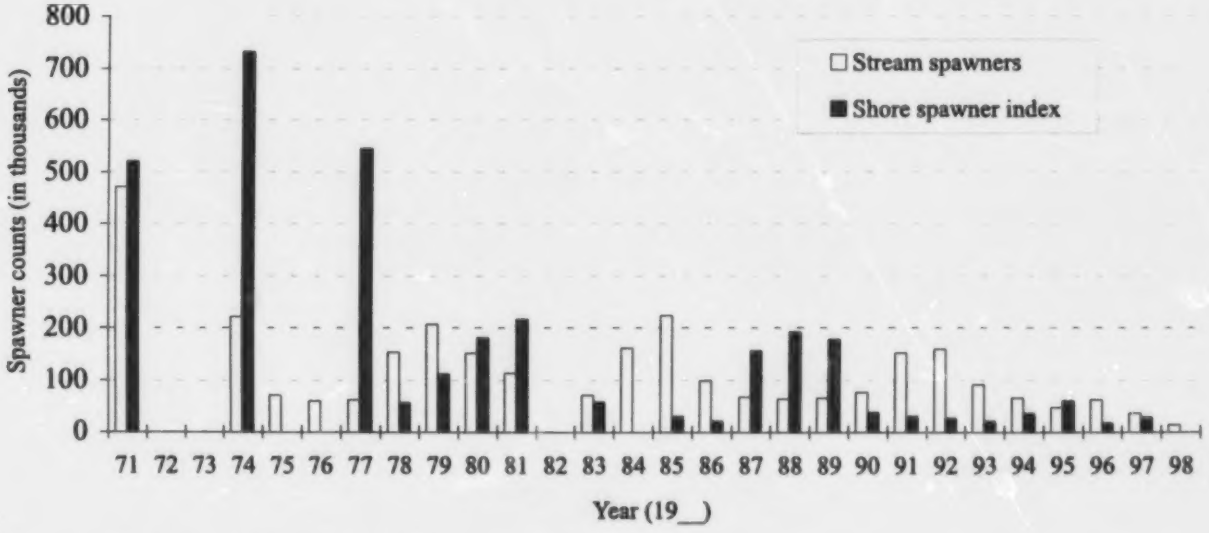

Figure 1. Okanagan Lake kokanee spawner enumerations, 1971–98.

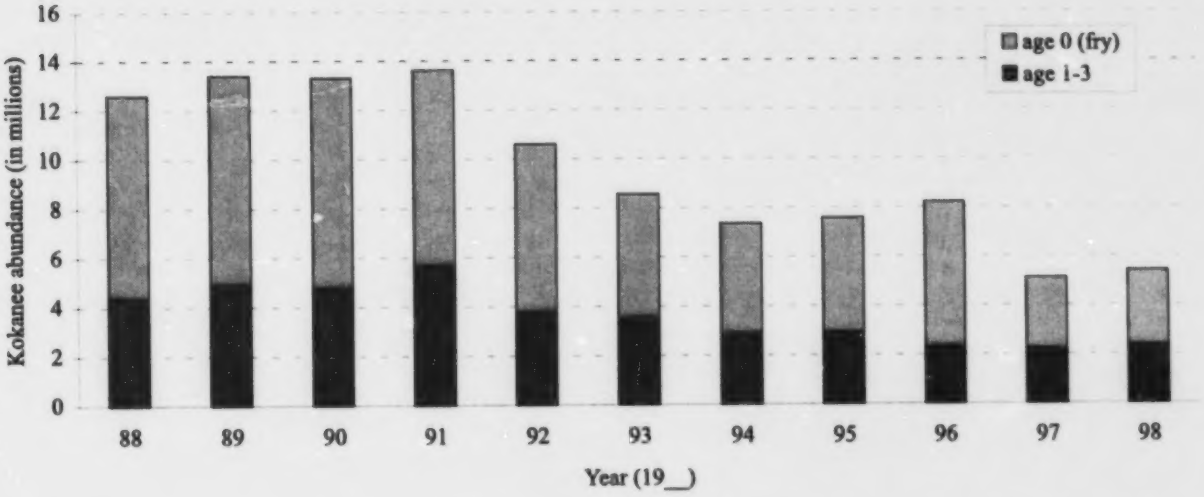

Figure 2. Index of kokanee abundance in Okanagan Lake based on hydroacoustic surveys, 1988–98 (data from D. Sebastian, British Columbia Ministry of Fisheries, Victoria, BC).

the typical salmonid tail-digging action. This behaviour draws males to the site, and they press into a "cluster" (Dill 1996) over the crevice that has been cleaned by the female. The female then passes through the cluster of males, and the combined group releases eggs and sperm to settle into the crevices. Spawner numbers peak around noon, and the spawning sites are generally abandoned by evening (Dill 1996).

Recent genetic studies using a variety of methods (Taylor

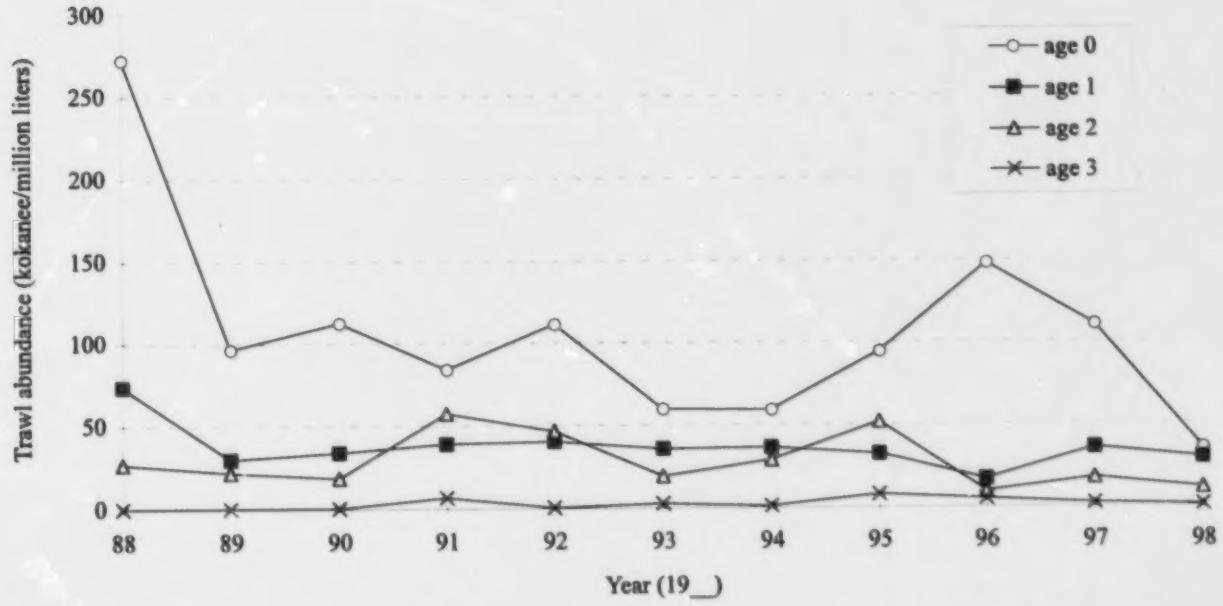

Figure 3. Index of kokanee abundance by age for Okanagan Lake based on trawl surveys, 1988–98 (data from D. Sebastian, British Columbia Ministry of Fisheries, Victoria, BC).

and Pollard 1998) suggest that the stream- and shore-spawning ecotypes of Okanagan Lake kokanee are closely related to each other and have a common ancestry. However, the results to date also indicate that gene flow is restricted between these 2 ecotypes, and improved microsatellite-assay techniques may allow strain identification and separation.

In the simpler days of dealing with just fisheries management concerns, stream-spawning stocks received the most attention, as it appeared that shore spawners contributed less to angler catches due to their smaller size at maturity (Shepherd 1990). Times, of course, have changed, and the first priority of the British Columbia Fisheries Program is now "the protection, maintenance and rehabilitation of native fish and their habitat to ensure ecosystem sustainability and biodiversity" (B.C. Fisheries Branch 1996). The unique biology of the shore-spawning kokanee of Okanagan Lake only adds to the argument for its protection and preservation together with the stream spawners.

WHY HAVE OKANAGAN LAKE KOKANEE STOCKS DECLINED?

So what has brought the Okanagan Lake kokanee runs to their knees, and what are we doing about it? The simultaneous decline of both the stream- and shore-spawning stocks without a compensatory increase in growth rates suggests that the primary problem has to do with a reduction in the lake's carrying capacity for kokanee (Ashley and Shepherd 1996). The primary cause of this reduction was hypothesized to be the introduction of the opossum shrimp, *Mysis relicta* (Shepherd 1996). Ironically, the introduction of *Mysis* into Okanagan Lake in 1966 was done with the intention of boosting the size of kokanee, based on a decade of experience with the introduction of *Mysis* into Kootenay Lake. As it turned out, *Mysis* were able to avoid predation by kokanee in the main basins of both lakes by sounding into the dark depths during the day. By virtue of their sheer numbers, together with their habit of coming up to feed all through the night on the same zooplankton species preferred by kokanee, *Mysis* instead became a formidable competitor. Conservative estimates of the current abundance of *Mysis* in Okanagan Lake are in excess of 62 x 10^9 animals, or 1,250 metric tonnes. That biomass is equivalent to well over 6 x 10^6 catchable-size (200 g) kokanee! Recent surveys revealed that dramatic autumn declines of *Mysis* in Armstrong Arm of Okanagan Lake were associated with significant increases in cladocerans, which are preferred prey of both kokanee and *Mysis* (McEachern 1999*a*). This response indicates that mysids may be controlling the abundance of other plankters important to kokanee.

Unfortunately, the Okanagan Lake kokanee stocks also face a number of other problems aside from *Mysis*, which complicate recovery efforts. Some parallels perhaps can be drawn between the Okanagan situation and the modelling exercise of Nickelson and Lawson (1998) for Oregon coho. In the case of Oregon coho, populations in reaches with poor incubation/rearing habitats became extinct during periods of low marine (for Okanagan kokanee, substitute in-lake?) survival; their probability of persistence increased with habitat quality.

Factors in the Decline of Stream-Spawning Stocks

The majority of the problems facing Okanagan kokanee stream-spawning stocks are associated with escalating human development pressures, set in a water-scarce situation. Of the 46 named tributaries to Okanagan Lake, most have seasonally ephemeral flows, and only 17 are known to support Okanagan Lake salmonid stocks. Of those known salmonid-producing streams, only 9 are currently utilized by kokanee. Humans were also attracted to the few year-round streams in this semi-arid region. Major irrigation developments began in the early 1900s, with detrimental consequences for fish; by the 1950s, >90% of the historic spawning areas in the main valley tributaries had been lost (Galbraith and Taylor 1969). Low summer-autumn flows further restrict the fish production potential of the remaining habitat in most of the tributaries to about half of optimum (Wild Stone Resources 1992).

For Mission Creek, the largest tributary to Okanagan Lake and historically the major kokanee producer of the Okanagan Lake tributaries, the situation is even worse. Up to 1983, Mission Creek carried an average of 85% of the lake's total returns of stream spawners; since then, the contribution of this stream has dropped disproportionately to an average of just over 50% (and <25% in 1998) of the total stream spawner returns. A host of perturbations of human origin probably factor in this decline, including upstream forestry and agricultural practices. In the lower reaches, urban encroachment has choked and channelled the floodplain; the dyking and dredging associated with the flood protection program has detrimentally impacted spawning habitat by siltation, scouring and outright removal of gravel.

Factors in the Decline of Shore-Spawning Stocks

Shore spawners have also been impacted by increasing human activities along the foreshore. Construction of piers and boat launches, retaining walls, breakwaters, and sand traps can change wave energy patterns and cause siltation of spawning habitats. Fines and other contaminants can also find their way to the foreshore as a result of a variety of upland activities. It is likely no accident that the reaches of Okanagan Lake most heavily utilized by shore spawners in recent years are areas that remain relatively free of foreshore development. But shore spawner numbers are down even in pristine stretches. While much of this decline is blamed on *Mysis*, it is probable that drops in lake levels for

flood protection purposes resulted in incubation losses of up to 50% in some years (Fig. 4; Shepherd and Sebastian 1998).

Other Factors in the Decline of Kokanee Stocks

In addition, there are other, larger forces at work in Okanagan Lake. A major program to control phosphorus discharges from sewage systems has operated in the Okanagan for almost 3 decades (Forty 1996), and levels of phosphorus declined steadily in Okanagan Lake from 1982 to 1994 (Bryan 1996). Decreasing nutrient loading would be expected to reduce the productive capacity of the lake for both *Mysis* and kokanee, and there are some indications that this is happening (Figs. 5 and 6; McEachern 1999*a*).

These in-lake declines have led some to suggest that a nutrient enrichment program should be considered for Okanagan Lake. Indeed, such a program has operated on Kootenay Lake for 6 years, with encouraging short-term boosts in kokanee production (Ashley et al. 1997). This strategy has not been pursued to date for Okanagan Lake, for at least 2 reasons. One reason is that the implementation of a nutrient addition program on the heels of major public investment in a long-term nutrient removal program obviously would appear to be inconsistent, and certainly is controversial from the socioeconomic perspective (Ashley and Shepherd 1996). A second and more technical concern is whether a nutrient enrichment program will work biologically in the longer term. While the results of the Kootenay fertilization experiment have favoured kokanee to date, it is pointed out that the total phosphorus target levels of 0.004–0.010 mg/L for Kootenay Lake (Ashley et al. 1997) are similar to or even less than the levels of total phosphorus presently in Okanagan Lake (McEachern 1999*b*), and spring total phosphorus levels have almost doubled in the 1995–98 period (V. Jensen, Pollution Prevention Branch, Penticton, unpubl. data.). Given that Okanagan Lake has kokanee productivity problems at these higher phosphorus levels, there is concern as to the long-term benefits of nutrient additions. However if kokanee numbers continue to drop, this may be a short-term approach worth considering to stave off extinction.

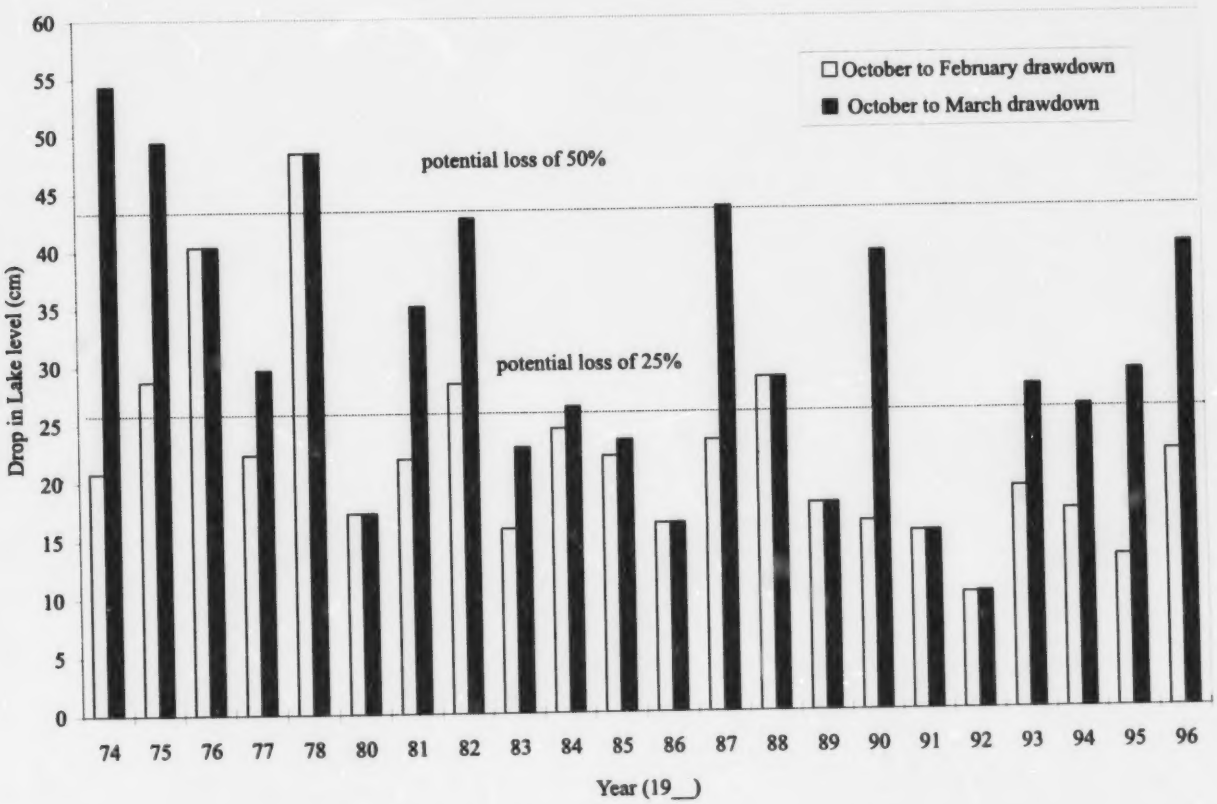

Figure 4. Predicted shore-spawning kokanee embryo losses due to drawdown of Okanagan Lake.

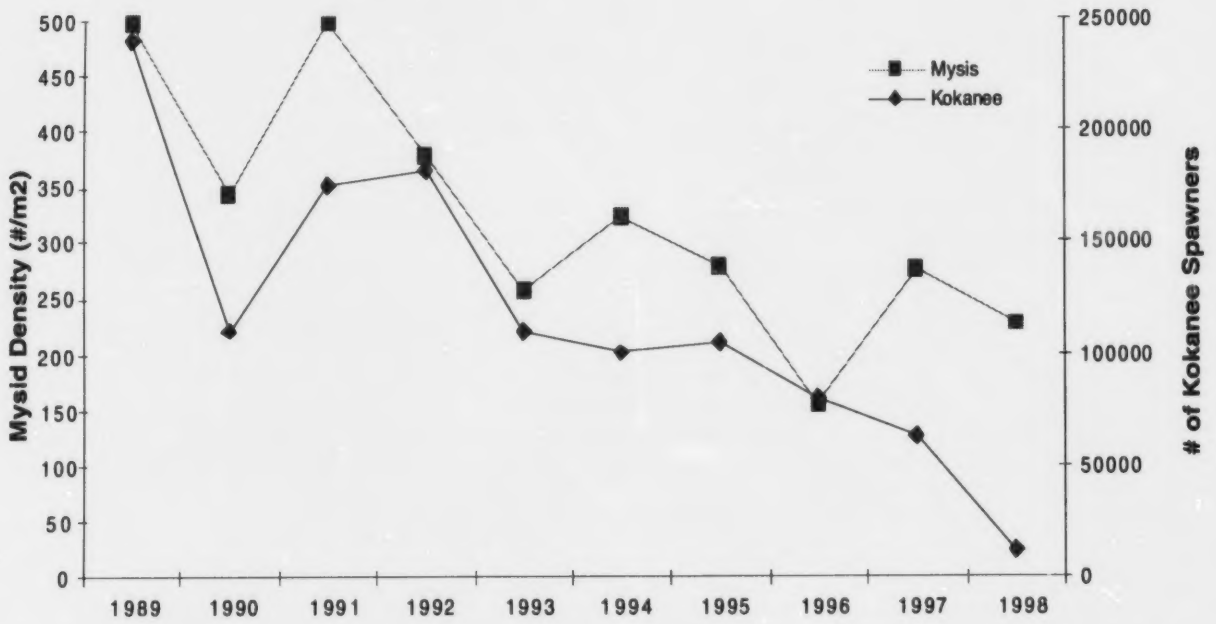

Figure 5. Trends in fall mysid abundance (vertical hauls) as compared with trends in numbers of kokanee spawners (fall counts) from 1989 through 1998, inclusive, in Okanagan Lake (from McEachern 1999*a*).

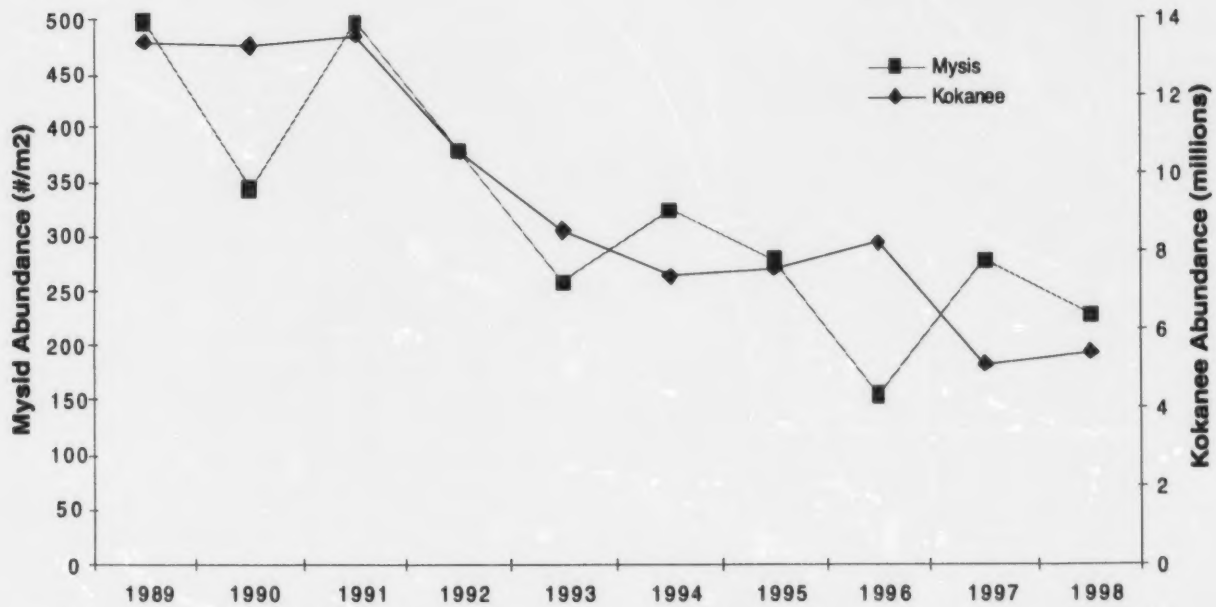

Figure 6. Trends in fall mysid abundance (vertical hauls) as compared with trends in kokanee abundance (hydroacoustic surveys) from 1989 through 1998, inclusive, in Okanagan Lake (from McEachern 1999*a*).

DEVELOPMENT OF THE OKANAGAN LAKE ACTION PLAN

In response to these declines, the British Columbia Ministry of Environment, Lands and Parks hosted an Okanagan Lake workshop in 1995, out of which developed a 20-year action plan to "rebuild and maintain the biodiversity of kokanee stocks (and other indigenous fish species) in Okanagan Lake" (Ashley and Shepherd 1996). The action plan was divided into 4 5-year phases, moving from conservation of remaining fish stocks and habitat, and collection of key information, through development and implementation of innovative resource management and restoration techniques. Action points identified for each phase were grouped into 6 general categories:

1. Priority Remedial Measures – actions to be undertaken as soon as possible to preserve the remaining kokanee stocks of Okanagan Lake.
2. Sustained Monitoring Program – aspects of Okanagan Lake ecology to be monitored on a regular and continuing basis in order to determine success or failure of initiatives.
3. Functional Studies – collection of data to fill critical information gaps leading to improved resource management.
4. Comparative Analyses – collection of information from other stocks, lakes, and time frames that by comparison may provide insights into the Okanagan Lake situation.
5. Large-scale Experiments – to explore the effects of various management options on the ecology of Okanagan Lake.
6. Long-term Applied Research – to examine feasibility of untested techniques that may offer significant benefits if successful.

Workshop participants agreed that some options should not be pursued at this time, including nutrient additions, hatchery stocking, and introductions of new species.

HIGHLIGHTS OF ACTION PLAN ACTIVITIES TO DATE

With the financial support of the Habitat Conservation Trust Fund, phase 1 of the action plan began in 1996, and will continue through to 2000. Many of the results discussed above were products of the phase 1 package. However for phase 1, emphasis was placed on: 1) the conservation of key fish habitats; and 2) the development of a practical means of *Mysis* control. Some highlights of the first 3 years work on Phase 1 include:

- improved database and understanding of the seasonal limnology (including *Mysis*) of Okanagan Lake, as well as Kalamalka and Wood lakes (adjacent "controls");
- better knowledge of kokanee shore spawner behaviour and habitat preferences;
- assessment of potential identification techniques for kokanee shore and stream stocks;
- production and distribution of fish habitat protection information packages to more than 5,000 water licensees and owners of waterfront properties, to encourage stewardship;
- increased profile and assistance of Fisheries staff in fish habitat protection cases;
- investigation into improving the rule curve used to draw down Okanagan Lake;
- feasibility of restoration of mainstem and off-channel habitats in Trout, Mission, Peachland and Trepanier creeks;
- assessment of sonic "thumpers" to selectively destroy *Mysis*, "bubble pumps" to entrain and loft *Mysis* into the upper water column, and pheromones to attract and trap *Mysis*;
- test fishing to evaluate feasibility and impacts of commercial harvest of *Mysis*.

IN CLOSING...

Humans continue to be attracted to the Okanagan, and development pressures are rising. Given these growing pressures, it will be a challenge to conserve both stream and foreshore habitats (especially those that are currently empty of fish due to other reasons) for the use of future fish. However, it is encouraging that many members of the public see the presence of kokanee as symbolic of ecosystem health, and an enhanced sense of public stewardship undoubtedly will be key in the perpetuation of Okanagan Lake kokanee stocks.

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Anatomy of a Population at Risk: Current Status of Okanagan River Sockeye (*Oncorhynchus nerka*) the Last Surviving Stock of Anadromous Salmon in Canadian Waters of the Columbia River Basin

K.D. Hyatt

Canada Department of Fisheries and Oceans, Science Branch
Stock Assessment Division, Pacific Biological Station, Nanaimo, BC, V9R 5K6, Canada
HyattK@pac.dfo-mpo.gc.ca

D.P. Rankin

Canada Department of Fisheries and Oceans
Science Branch, Stock Assessment Division,
Pacific Biological Station, Nanaimo, BC, V9R 5K6, Canada

J. Fryer

Columbia River Intertribal Fish Commission
200-729 NE Oregon Ave., Portland, OR 97232 USA

D. Macchin

Okanagan Nation Fisheries Commission,
Okanagan Nation Alliance
3255C Shannon Lake Rd.
Westbank, BC, V4T 1V4, Canada

B. Hanslit

Canada Department of Fisheries and Oceans,
Science Branch, Stock Assessment Division,
Pacific Biological Station
Nanaimo, BC, V9R 5K6, Canada

M. Johannes

Northwest Ecosystem Institute
Lantzville, BC, V0R 2H0, Canada

ABSTRACT

Prior to the early 1900s anadromous salmon populations of several species occupied watersheds throughout portions of the Columbia River Basin in both Canada and the United States. Development of dams on the Columbia River and its tributaries to manage water for hydroelectric power generation, flood control and irrigation purposes has resulted in the elimination of many but not all salmon populations returning to upstream portions of the basin. In Canadian portions of the Columbia River Basin, all populations of several species of salmon were extirpated with the notable exception of Okanagan River sockeye salmon (*O. nerka*). Although not currently listed as endangered, a host of risk factors operate at each of several life history stages to threaten the long-term survival of the last of Canada's southern transboundary salmon stocks. Examination of growth and survival schedules of Okanagan sockeye in recent decades indicate that incubation and rearing conditions in the Okanagan River and Osoyoos Lake, respectively, still support above average levels of annual production. By contrast, time series observations of both survival and migratory behaviour indicate that riverine and ocean survival have been unfavourable enough in many recent years to cause steep declines in population size. Further, recent research indicates that Okanagan sockeye are routinely subjected to conditions at or beyond the thresholds normally considered to be appropriate for this species at all life history stages from spawning adults, through incubating eggs, to lake and ocean rearing subadults. Given these observations and projections that Canada's Okanagan Valley will continue to develop rapidly as the third major urban area in British Columbia, we conclude that Okanagan sockeye salmon are clearly at risk of loss in the foreseeable future. To avoid this, all levels of government will need to adopt an ecosystemic approach to land, water and other resource management practices in order to increase the probability of long term persistence of Okanagan sockeye.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

The Biology and Management of the Salish Sucker and Nooksack Dace

Mike Pearson

Westwater Research Centre, Institute for Resources and Environment
Level 4, 2206 East Mall, University of British Columbia, Vancouver, BC, V6T 1Z3, Canada
mpearson@portal.ca

ABSTRACT

Salish sucker (*Catostomus* sp.) and Nooksack dace (*Rhinichthys* sp.) are undescribed and endangered members of a unique assemblage of freshwater fish that evolved in an isolated refuge in Washington State during the Pleistocene glaciations. Canadian populations are limited to a few headwater streams in the lower Fraser Valley, B.C. and are in rapid decline. This report summarizes existing knowledge of the life history and probable causes of decline of both species, and discusses research needs and management options for their conservation. Loss of riffle habitats and decreased in-stream habitat complexity are likely factors in population decline. Habitat fragmentation, sublethal temperature effects, and interactions with exotic species may also be important. Key gaps in knowledge important for management include winter habitat requirements, spawning site locations, habitat fragmentation effects, thermal tolerances, and the identification of effective habitat restoration techniques. Development of appropriate municipal policies for habitat protection are critical due to the limited spatial distribution of both species and the prevalence of land-use related causes of habitat degradation. Increased interest in these species and rising commitment to conservation of biodiversity among government agencies, municipalities, stewardship groups, and the general public provide some grounds for optimism for the survival of these species in Canada.

Key words: *Catostomus*, Chehalis fauna, Fraser Valley, Nooksack dace, *Rhinichthys*, Salish sucker, Washington State.

Salish sucker (*Catostomus* sp.) and Nooksack dace (*Rhinichthys* sp.) are the only Canadian representatives of the Chehalis fauna, a unique fish community that survived continental glaciation in an ice-free refuge in Washington State (McPhail 1967). They are nationally and provincially endangered (Campbell 1990, Cannings 1992, McPhail 1997), and within Canada are found only in a handful of headwater streams in the lower Fraser Valley, B.C. Populations are declining rapidly due to habitat loss caused by urbanization, aggregate extraction, and flood-control dredging (McPhail 1987, 1997; Pearson 1998a).

Although the initial description of the sucker is more than 50 years old (Schultz 1947), until this decade only McPhail (1967, 1987, 1997) had taken an active interest in either species. Since 1990, the British Columbia Ministry of Environment, Lands and Parks (MELP) has conducted a series of studies on distribution and habitat preferences (Inglis et al. 1992, 1994), population status (McAdam 1995), and habitat availability (Pearson 1998a,b). This research is continuing through the Westwater Research Centre of the University of British Columbia.

In this paper I review what is known of the origins, distribution, and life history of both species and discuss possible causes of population decline and the challenges of managing species at risk in 1 of the most rapidly developing regions in Canada.

ORIGINS

The Salish sucker and Nooksack dace recently diverged from 2 common and widespread species, the longnose sucker (*Catostomus catostomus*) and the longnose dace (*Rhinichthys cataractae*) respectively. As the range of these parental species contracted with the onset of glaciation, the fish of the Chehalis Valley were left as peripherally isolated populations (McPhail and Taylor 1996). The valley remained ice-free through all 4 major glaciations of the Pleistocene (McPhail 1967, McPhail and Lindsey 1986). Recent genetic work indicates that both species[1] have been reproductively isolated since well before the most recent glacial episode and perhaps since before the Pleistocene (McPhail 1997).

As the ice sheets withdrew, the Chehalis fauna (a number of other species share this history) dispersed into the newly exposed landscape. Salish sucker and Nooksack dace ventured

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

further north than the others, eventually establishing themselves in the lower Fraser Valley of Canada. Their route was presumably via the temporarily freshwater Puget Sound and a series of lakes that filled the gap between the ice and the Coastal Mountains during the glacial retreat (Thorson 1980). The suckers and dace were likely among the very first species to recolonize the post-glacial streams of the lower Fraser Valley (McPhail and Carveth 1993).

DISTRIBUTION

Nooksack dace currently inhabit streams on the east side of Puget Sound and on the west side of Washington's Olympic Peninsula (Fig. 1). Oddly, they are absent from the drainages lying between these regions on the west side of Puget Sound (McPhail 1997). In Canada their distribution is restricted to tributaries of the Nooksack River: Bertrand, Cave, Pepin, and Fishtrap creeks (Inglis et al. 1994, Pearson 1998*a*).

Salish suckers are known from 6 river systems of the Puget Sound Lowlands and the lower Fraser Valley (Fig. 1). These are: the lower Fraser (Salmon and Salwein rivers, and Semiault Creek); the Little Campbell River; the Nooksack system (Bertrand, Cave, Pepin, and Fishtrap creeks, and Whatcom Lake); the Stilliguamish drainage (Twin Lakes); the Green River; and Lake Cushman of the Skokomish system (McPhail and Taylor 1996).

Figure 2 summarizes the current distribution of Salish sucker and Nooksack dace in Canada from sampling records since 1990. More detailed maps and data are provided by Pearson (1998*a*). It is important to remember in reviewing this presence/absence data, that fish density varies widely among sites and that, with very few exceptions (see below), those of suckers and dace are very low. In addition, given their rarity, the failure to collect a species at a given site does not necessarily imply absence.

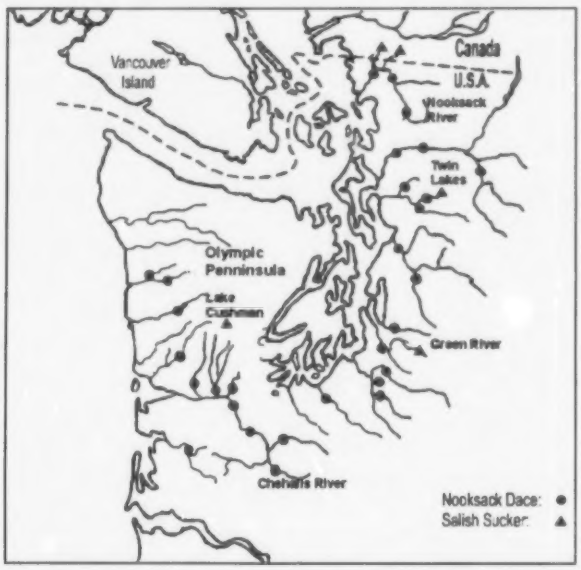

Figure 1. Current global distribution of Salish sucker (▲), Catostomus sp., and Nooksack dace (●), *Rhinichthys* sp. (adapted from McPhail 1997).

POPULATION TRENDS AND CONSERVATION STATUS

Neither species appears imminently threatened in Washington State. Two of the 4 sucker populations are located within protected areas or buffers (J. McPhail, University of British Columbia, pers. comm.), and the dace remain common in most of their native streams (McPhail 1997). In contrast, both species are in rapid decline within Canada. The Salish sucker is considered extirpated from the Little Campbell River (circa 1976; McPhail 1987) and perhaps from the Salwein River (Inglis et al. 1992). Both species have also disappeared from Howes Creek, a tributary of Bertrand

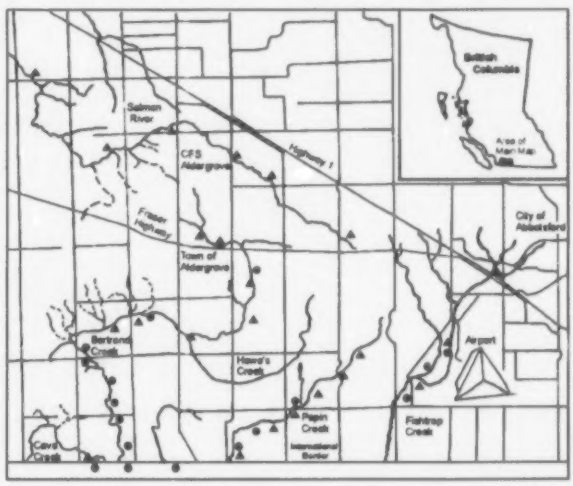

Figure 2. Current Canadian range of Salish sucker (▲) and Nooksack dace (●). A small population of suckers is also known from Semiault Creek in Chilliwack (40 km west of mapped area). The Little Campbell River, from which suckers are believed extirpated, is immediately west of the mapped area.

[1] Although the taxonomic status of neither Nooksack dace nor Salish sucker is established, there is an excellent case for granting species status to both based on their genetic and morphometric uniqueness (McPhail and Taylor 1996, McPhail 1997). I will refer to them as species in this paper for reasons of convenience.

Creek (Inglis et al. 1992, 1994). Dace remain abundant only in Bertrand Creek south of 16th Avenue (Inglis et al. 1994, Pearson 1998*a*), and only lower Pepin Creek and perhaps the upper Salmon River contain healthy numbers of Salish sucker (Inglis et al. 1992, Pearson 1998*a*).

Declines are almost certainly due to progressive habitat loss. The human population of the lower Fraser Valley is 1 of the most rapidly growing in the country. Riffle habitat is quickly disappearing through ponding, siltation, and dredging. Summer flows are decreasing or stopping altogether as surface runoff associated with deforestation, urbanization, and agricultural drainage increases. Dace, for example, were very common in Fishtrap Creek south of Echo Road until virtually all the riffle habitat was eliminated by municipal dredging for flood control between 1989 and 1991, and headwater regions of the Salmon River that once supported healthy populations of sucker now run dry every summer (J. McPhail pers. comm.). The situation of the Canadian populations is exacerbated by their isolation. The dace are separated from the Washington Nooksack River populations by severely degraded habitat in the American portions of Bertrand and Fishtrap creeks (McPhail 1997), and the nearest sucker populations lie 100 km and 2 drainage basins to the south (McPhail and Carveth 1993).

Both species are Red-listed in British Columbia (Cannings 1992) and designated Endangered by the federal Committee on the Status of Endangered Wildlife in Canada (COSEWIC; Campbell 1990, McPhail 1997). Neither is protected in the United States.

LIFE HISTORY

Nooksack Dace

Nooksack dace are benthic riffle specialists as adults. Inglis et al. (1994) used multivariate regressions to generate habitat suitability curves. They found that adults prefer water depths of 10–19 cm at velocities of 25–30cm/sec over gravel, cobble, or small boulders. Winter habitat use has not been well studied. McPhail (1997) suggests that they probably inhabit riffles year-round in the Fraser Valley, but may shift to pools in more severe environments. The author has collected adults from cobble riffles in Bertrand Creek during November and January.

Dace spawn at night during April and May near the upstream end of riffles, with females depositing 200–2,000 eggs depending on body size (McPhail 1997). There is no information on courtship, parental behaviour, or incubation periods. Young-of-the-year dace aggregate in shallow, marginal pools over mud or sand substrates near the downstream ends of riffles. They feed on chironomid larvae and ostracods (McPhail 1997). Aggregations break up in late summer (J. McPhail pers. comm.), and have been observed as late as 8 September by the author.

Both sexes reach maturity at the end of their second summer and spawn for the first time in their third spring. There are generally 4 year-classes present (Inglis et al. 1994) and the oldest known individual was a female in her sixth summer (McPhail 1997). Individuals exceeding 100 mm in fork length are rare.

The few gut contents examined indicate that adult dace feed primarily on riffle-dwelling insects, including caddisfly and mayfly nymphs, dytiscid beetle larvae, and adult riffle beetles (McPhail 1997). They have been observed foraging both at night (J. McPhail pers. comm.; Pearson 1998*a*) and during daylight hours (Pearson 1998*a*). Individuals collected at mid-morning have empty stomachs, but full intestines, suggesting that feeding is nocturnal (McPhail 1997).

Nooksack dace typically occur with cutthroat trout (*Oncorhynchus clarki*), prickly sculpin (*Cottus asper*), rainbow trout (*O. mykiss*), juvenile steelhead (sea-run *O. mykiss*), and juvenile coho salmon (*O. kisutch*), all of which are potential predators. Indeed, cutthroat have been observed feeding on young dace (Inglis et al. 1994).

Salish Sucker

Adult Salish suckers use a variety of habitat types. In Canada they are found in small headwater streams and associated ponds; in Washington, however, several lake populations exist (McPhail 1987). They are caught in a variety of water velocities and depths, but are most often found in slow currents over sand or silt substrate in areas with in-stream vegetation and over-stream cover (Inglis et al. 1992). Young-of-the-year Salish suckers are found in habitats similar to those used by adults, but seem to prefer more overhanging vegetation (Inglis et al. 1992). Winter habitat remains unknown, but it seems likely that they require off-channel refuge to escape from the frequent high flows associated with winter rains in Fraser Valley creeks.

Suckers spawn in riffles over fine gravel at current velocities of up to 50 cm/sec (McPhail and Taylor 1996) beginning in March or April, when water temperatures reach 7–8°C (McPhail 1987). The period is very protracted and individuals in spawning condition have been captured throughout the summer, even in late July at water temperatures in excess of 20°C (Inglis et al. 1992, McAdam 1995, McPhail and Taylor 1996). Fecundity is unknown, but preserved females contain "large numbers of small eggs" (McPhail 1987). Like other species in the genus, Salish suckers are broadcast spawners. No nest is built and the adhesive eggs stick to gravel and rocks. Those on top are usually consumed by predators, but the current carries many under gravel and cobble where they are more protected (J. McPhail pers. comm.). No information is available on incubation period.

Only 2 current spawning sites are known: Pepin Creek at 0 Avenue, and Pepin Creek at Bradner Road. Previously used sites include the Salmon River at 256th, 272nd, and 280th

streets, and at 40th and 48th avenues, but these have not been monitored in more than 20 years, and most have been severely degraded since then (J. McPhail pers. comm.). Recruitment appears sporadic and is effectively zero in many years (J. McPhail pers. comm.).

There appear to be 5 year-classes in British Columbia populations (McPhail 1987, Inglis et al. 1992) although older individuals are known from Washington (McPhail 1987). Males are sexually mature in their second year and females in their third year, with the minimum size of spawners being 87 mm for males and 95 mm for females (McPhail and Taylor 1996). The largest individual known from Canadian waters (244 mm fork length) was captured in Pepin Creek in 1992 (Inglis et al. 1992).

Dietary information is limited to gut content analysis of 10 adults, all of which contained primarily detritus and large numbers of chironomid head capsules. The diet of the young is unknown (McPhail 1987). Salish sucker fry and juveniles are probably preyed upon by many of the species with which they share their habitat, including various salmonids, prickly sculpin, and introduced species (see below).

POSSIBLE FACTORS LIMITING POPULATIONS

Populations may be limited by a variety of factors, both abiotic and biotic. These include loss of physical habitat, elevated temperatures, water quality degradation, changes in flow regime, competition, predation, and food. Given the magnitude of alteration and degradation these streams have endured over the past 200 years (Healey 1997), it seems likely that a number of factors are acting in concert to limit populations. The following is a brief discussion of some likely candidates.

Loss of Riffle Habitat

Dace spend virtually their entire adult lives in riffles, and both suckers and dace require gravel or cobble riffles for spawning. A large proportion of riffle in their native streams has been lost to the dredging, siltation, and ponding associated with urbanization, agricultural drainage, and aggregate extraction operations. Increased runoff rates (and consequent lack of groundwater recharge) caused by these developments have also reduced summer discharge levels in many reaches. Surface flow ceases for up to 2 months during most summers in Cave Creek, the upper Salmon River, and many small tributaries across the species range (Pearson 1998*b*). This, in effect, eliminates riffle habitat at the most productive time of year for dace (McPhail 1997).

Loss of In-stream and Over-stream Cover

Channel simplification and loss of riparian vegetation has impacted many reaches, particularly in pasture lands and urban areas. Inglis et al. (1992) demonstrated a marked preference for such cover in suckers, particularly juveniles. Many reaches also lack significant off-channel habitat, which may provide important refuge areas during high flows, particularly for the suckers.

Stream Temperatures/Low Oxygen

Temperature preferences and lethal limits for Nooksack dace and Salish sucker remain unknown, but circumstantial evidence suggests that both species can survive at least short-term exposure to relatively high temperatures (Pearson 1998*a*). Although acute mortality of suckers or dace is unlikely in most reaches at the present time, there are localized areas of concern (Pearson 1998*b*). Sublethal effects, including reduced growth and fecundity and increased susceptibility to disease may also be important.

Hypoxia is commonly associated with high water temperatures and nutrient loading, both of which affect numerous reaches within the study area during late summer (Pearson 1998*b*).

Habitat Fragmentation

All of the factors listed above are likely to reduce populations by decreasing the total area of suitable habitat. Their spatial distribution may also, however, have major impacts by restricting movements and dispersal of suckers and dace, at least seasonally. In addition, dams and poorly designed culverts can block fish passage under some conditions. The effects of such anthropogenic habitat fragmentation on fish is virtually unstudied, but likely to be very important. Many studies of terrestrial habitat fragmentation, which has been a major research focus in conservation biology over the past decade, have demonstrated that habitat fragmentation can accelerate rates of population declines over those produced by area loss alone (Saunders et al. 1991, Andren 1994, Gonzalez et al. 1998). Organisms near the edges of fragments are also exposed to influences from the adjacent hostile habitat. Although such "edge effects" are generally accepted to be deleterious, there is little agreement on definitions, how harmful they are, or even how to measure them (Murcia 1995).

Two key attributes of streams are likely to alter their responses to fragmentation relative to those of terrestrial habitats: streams are linear and directional. In a linear system, fragmentation will proceed much faster than in a 2-dimensional landscape, particularly in the initial stages of habitat destruction. Unlike a 2-dimensional area, in which habitat loss initially appears as holes in intact habitat and separate patches do not form until a considerable amount of habitat is lost (Andren 1994), each unit destroyed in a linear system will either subdivide existing habitat or add to the distance between patches. Consequently the effects of reduced patch size and increased isolation are likely to appear when a

Table 1. Introduced species reported from the current Canadian range of Salish sucker and Nooksack dace (from Pearson 1998*a*).

Creek	Species	First recorded
Fishtrap Creek	Pumpkinseed (*Lepomis gibbosus*)	1995
	Black crappie (*Pomoxis nigromaculatus*)	1997
	Largemouth bass (*Micropterus salmoides*)	1997
Cave Creek	Pumpkinseed	1997
	Fathead minnow (*Pimephales promelas*)	1992
Pepin Creek	Largemouth bass	1997
Bertrand Creek	Black bullhead (*Ameiurus melas*)	1997

much smaller proportion of total habitat is destroyed than in a terrestrial system. The directional nature of stream habitats will influence edge effects greatly. For example, alterations in temperature, sediment load, and large woody debris input levels will have exaggerated impacts on downstream habitats and leave upstream habitats relatively untouched.

Reaches in which habitat has been destroyed or severely altered are unlikely to be completely hostile at all times. Seasonal changes in flow, temperature, and water quality will alter the pattern and distribution of fragmentation. A critical period for Salish sucker and Nooksack dace is likely to occur in late summer, when thermal, hypoxia, and low-flow related barriers are all most likely to occur. Currently, too little is known of movement timing or extent in either species to assess likely impacts. Other studies have shown that species with generalized habitat requirements, like the Salish sucker, are more likely to either tolerate changes within the patch or make use of "degraded" habitat than are specialists such as the dace (Andren 1992, Diffendorfer et al. 1995, Sarre 1995). Species with relatively large home ranges may also simply use several patches by moving between them, perhaps expanding the home range to accommodate habitat loss (Rolstad 1991).

Introduced Species

Exotic species are becoming established in the Nooksack tributaries in increasing numbers (Table 1), and may impact sucker and dace populations either competitively or through direct predation. Several of the introduced species are at least partially piscivorous.

RESEARCH AND MANAGEMENT NEEDS

Without significant habitat protection and restoration measures, the likelihood of the Canadian populations of Salish sucker and Nooksack dace surviving more than a few decades seems slim. Those remaining are already perilously close to extirpation, and what occurs over the next 5–10 years is likely to determine their fate. Specifically, spawning, rearing, and overwintering habitat needs to be identified and protected. Stream temperatures should be at least stabilized at current levels, and sources of siltation should be identified and reduced. In the longer term, changes in the hydrograph stemming from urbanization need to be stopped and hopefully reversed—especially decreased summer flows. All of this will require a high level of commitment and cooperation from landowners, government agencies, and local stewardship groups.

Municipal governments in particular are critical to successful conservation efforts for these and many other species at risk in heavily settled areas. A large proportion of threatened and endangered species have very restricted geographic ranges, which correspond most closely to local government jurisdictions in spatial scale. Furthermore, the majority of threats to species in these areas relate to altered land uses, which are largely under municipal control (Press et al. 1996). Powerful policy tools, including zoning by-laws, official community plans, designation of environmentally sensitive areas, and tree by-laws are available to local governments. With the necessary political commitment and some imaginative application these could become major contributors to biodiversity preservation in British Columbia. Indeed, I would argue that without their application there is little long-term hope for many species at risk, including the Salish sucker and Nooksack dace.

Fortunately, there are considerable grounds for optimism on this front. Of the 2 municipalities that contain virtually all of the Canadian range of Salish sucker and Nooksack dace, one (Abbotsford) has hired a Stream Stewardship Coordinator and has established a roundtable committee to support her work. The other (Langley) has an Environmental Coordinator and works closely with the Langley Environmental Partners Society on stream rehabilitation work. A number of local stewardship groups are active in habitat restoration. The necessary ingredients for effective action seem to be falling into place. The long-term survival of these unique species will provide a very simple test of our ability to sustainably manage watersheds with multiple and competing land uses and values.

ACKNOWLEDGEMENTS

A number of people provided invaluable assistance to this work. T. Patton was an enthusiastic and competent field assistant. K. Payette of MELP Surrey, fulfilled my many requests for maps, aerial photos, and files. J. Rosenfeld of MELP Fisheries Research, and my research supervisor, M. Healey, contributed plenty of good advice, and J. McPhail

gave access to his encyclopedic knowledge of the area's fish fauna. This research is financially supported by British Columbia's Habitat Conservation Trust Fund.

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Conserving Genetic Diversity in Rainbow Trout

Eric Parkinson
British Columbia Fisheries Branch, 2204 Main Mall, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada
Eric.Parkinson@gems9.gov.bc.ca

Eric Taylor
Department of Zoology, University of British Columbia
Vancouver, BC, V6T 1Z4, Canada

Ernest Keeley
British Columbia Fisheries Branch
2204 Main Mall, University of British Columbia
Vancouver, BC V6T 1Z4, Canada

ABSTRACT

Conserving genetic diversity is a major concern for north temperate fauna which are species poor but rich in genetic diversity. The problem is believed to be especially serious for freshwater fish where populations are isolated by barriers to migration. Isolation and low species diversity have been hypothesized to be the key ingredients in an adaptive radiation that has taken place in British Columbia fish since the last glaciation. Our work documents this variation and produces a within-species taxonomy of rainbow trout as a template for other species. Our work also incorporates both molecular data (presumably selectively neutral) as well as morphological and life history data (presumably adaptive). Molecular data are being used to identify ancient subdivisions within the species and patterns of gene flow among interconnected populations. Morphological and life history data will be used to group isolated populations into common adaptive ecotypes. The output of this analysis will be linked to an electronic atlas of British Columbia and will be used to predict the locations and characteristics of rare and threatened genotypes.

Key words: genetic diversity, genotype, population structure, rainbow trout, taxonomy.

In species-rich situations, conservation efforts have typically focused on species or communities of species as the elements that need to be conserved. However, many species vary substantially among populations, in addition to species or subspecies with Linnean nomenclature. The problem is particularly acute in freshwater fish where barriers to movement can subdivide species into hundreds or thousands of populations, commonly called stocks in the fisheries literature (Larkin 1972). These stocks range from semi-isolated populations with independent population dynamics, but little genetic differentiation, to completely isolated populations that are uniquely adapted to local environmental conditions (Taylor 1991). In north temperate areas such as British Columbia, the problem is particularly acute because barriers to movement have combined with climatic and geological diversity to produce ideal conditions for speciation among the relatively few fish species that invaded these areas following the last glaciation. Many examples exist of local adaptations that verge on species-level differences which are confined to a single watershed or habitat type (Taylor 1991). These subdivisions do not fit the traditional concepts of subspecies in terms of the number of different phenotypes represented and the lack of phylogenetic and geographic contiguity for populations with similar phenotypes.

This issue is acknowledged in the U.S. Endangered Species Act which recognizes the status of both subspecies and "distinct population units." Consequently, the concept of an Evolutionarily Significant Unit (ESU) was developed to describe "distinct population units" in Pacific salmon (Waples 1995). The practical problem in conservation is to determine the number of ESUs and then to assess their geographic location, distinctiveness, rarity, and risk of extinction. Conservation efforts can then be directed to sites where distinctive, rare ecotypes are at high risk of extinction.

Several approaches to defining ESUs are available. Individual populations can be assessed to determine if they meet the criteria in the original definition (Waples 1995). Usually this approach is applied to populations that are perceived to be of adaptive significance and appear to be endangered. However, unique populations without obvious distinguishing characteristics can easily be overlooked. Another approach is to divide the entire species into contiguous geographic blocks based on molecular genetic affiliations and general life history characteristics. This approach is based on the assumption that these blocks represent

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

monophyletic groups with little internal genetic variation. A third approach, and the one we propose to pursue, involves using molecular genetics to define geographically contiguous groups that are presumably monophyletic. Within these monophyletic groups, however, each population is assigned to a particular ESU based on a suite of habitat and phenotypic characteristics in a system similar to that proposed by Bernatchez (1995). Our knowledge will be imperfect and we will make errors in classifying individual populations. However, the status of each population will be systematically assessed against the diversity found in the entire species to produce the best possible picture using currently available data. Information concerning the reproductive isolation of populations is important in this approach, as well as the others, because reproductive isolation is a basic criteria for defining an ESU.

Our assessment of the diversity within species consists of 3 basic tasks: 1) defining the smallest possible groups of monophyletic populations; 2) developing rules to define the population units within the monophyletic groups in terms of their reproductive isolation; and 3) classifying isolated (or semi-isolated) populations within monophyletic groups based on presumed adaptive differentiation to assess the biological uniqueness of each population. This methodology is being applied to define the structure of rainbow trout as a prototype for other species. Rainbow trout are clearly polytypic and an ample body of ecological knowledge as well as data on molecular and phenotypic variation exists. In addition, all legal fish introductions are recorded in a well-organized database, and, for large areas of British Columbia, there have been few official introductions and little possibility of illegal introduction.

The output of the within-species taxonomy will be tied to the British Columbia Watershed Atlas (http://www.elp.gov.bc.ca/fsh/ids/gis/). The Watershed Atlas is a geographic information system (GIS) at 1:50,000 scale which has stream reaches and lakes as its basic elements (macroreaches). Stock characteristics will be assigned to >1 million macroreaches. Reaches that share a common stock will share a common stock number; those that share a common phylogeny will share a common phylogenetic code; and those that share a common ecotype will share a common ecotype code. Preliminary estimates suggest that there are thousands of stocks, <10 phylogenetic groups, and <100 ecotypes. The output can be displayed in a database or in a map. Revisions in the system will result in new GIS layers for the Watershed Atlas with older classification systems retained for comparison.

Defining Monophyletic Groups within Species

A monophyletic group of populations shares a common ancestral population. A species can be envisioned as a hierarchy of monophyletic groups where upper levels in the hierarchy represent older subdivisions within the species. Monophyletic groups can be recognized by the similarity of molecular markers among isolated, but geographically contiguous populations. This technique has a well-established theoretical basis (e.g., Bernatchez 1995), but the number of phylogenetic groups, their boundaries, and the number of hierarchical levels within these phylogenies has not been established for any fish species in British Columbia.

In British Columbia our main interest is in identifying phylogenetic groups that invaded the province from different refugia following glaciation. This type of split is evident in British Columbia fish species. Rainbow trout are clearly distinguished into an interior and a coastal form by isozyme frequency differences (Parkinson 1984). In addition, some evidence indicates a north–south split on the coast. Other species (longnose sucker, longnose dace) also appear to have coastal and interior forms suggesting that differentiation in glacial refugia is a general phenomenon that forms an important component of the diversity within many British Columbia fish species.

From our understanding of the number of glacial refugia and invasion routes following glaciation, the number of groups originating from distinct glacial refugia must be <10. The key indicators of phylogenetic grouping are molecular genetic characteristics that are uniform within, but differ among, large geographic areas. As a result, a few samples covering a large geographic area can outline the phylogenetic structure. Additional samples taken from boundary areas provide information on the location and sharpness of boundaries. Even with this simplification, a substantial effort is required to adequately document the structure of each species. In extending this concept to other species, an expedient approach is to use whatever information is available until more definitive genetic data are collected. Useful data include phylogenetic boundaries of other species, barriers to fish dispersal, biogeographic patterns in fish communities (which indicate ancient dispersal barriers), glacial history, and morphology data.

Although this methodology is well established, there are obvious difficulties attempting to quantify genetic variation by sampling every population of a species in an area the size of British Columbia. Usually, the pattern of variation must be inferred from incomplete information. Our current understanding of the phylogenetic structure in rainbow trout is outlined in Fig. 1.

Defining Population Structure within Phylogenetic Groups

Populations rarely exist in isolation (i.e., total lack of gene flow from each other). Many fish, and salmonids in particular, are known for their long distance migrations and occasional straying. Even populations totally isolated above impassable barriers have a temporal element; the population

may not have been isolated long enough for genetic differentiation to occur. For cultured species such as rainbow trout, an introduction in a single lake may introduce foreign genes into a variety of populations within an interconnected watershed. An accurate picture of the population structure in a species such as rainbow trout must reflect the variation in genetic exchange rates among proposed populations.

In theory, 3 regimes can be envisioned. At very high rates of gene flow, even very strong selective forces do not induce genetic differentiation. At intermediate rates, lower rates enable increasingly weak selective forces to induce differentiation. At very low rates, neutral characters diverge over time because no selection is needed to induce differentiation. Molecular genetic data can only capture the lower end of this continuum. Evidence of population structuring in less isolated populations will be forced to rely on other evidence such as movement patterns or phenotypic differences.

Population structuring is probably related to the geographic structure of the environment. Some populations are isolated by physical barriers. Populations that are farther apart are more isolated. Migration and dispersal patterns differ among populations in different environments. If strays are more successful in similar habitats, then populations in dissimilar habitats should experience lower rates of gene exchange. Consequently, the genetic structure of the species will be related to the geographic structure of the habitat that it occupies.

Linkages between populations occupying interconnected water bodies can be assessed using data on molecular variation combined with data on the patterns of fish movement. Because a species can consist of thousands of populations, patterns of movement and genetic variation can be observed only in representative situations. However, if habitat data related to barriers to dispersal are available, then the likelihood

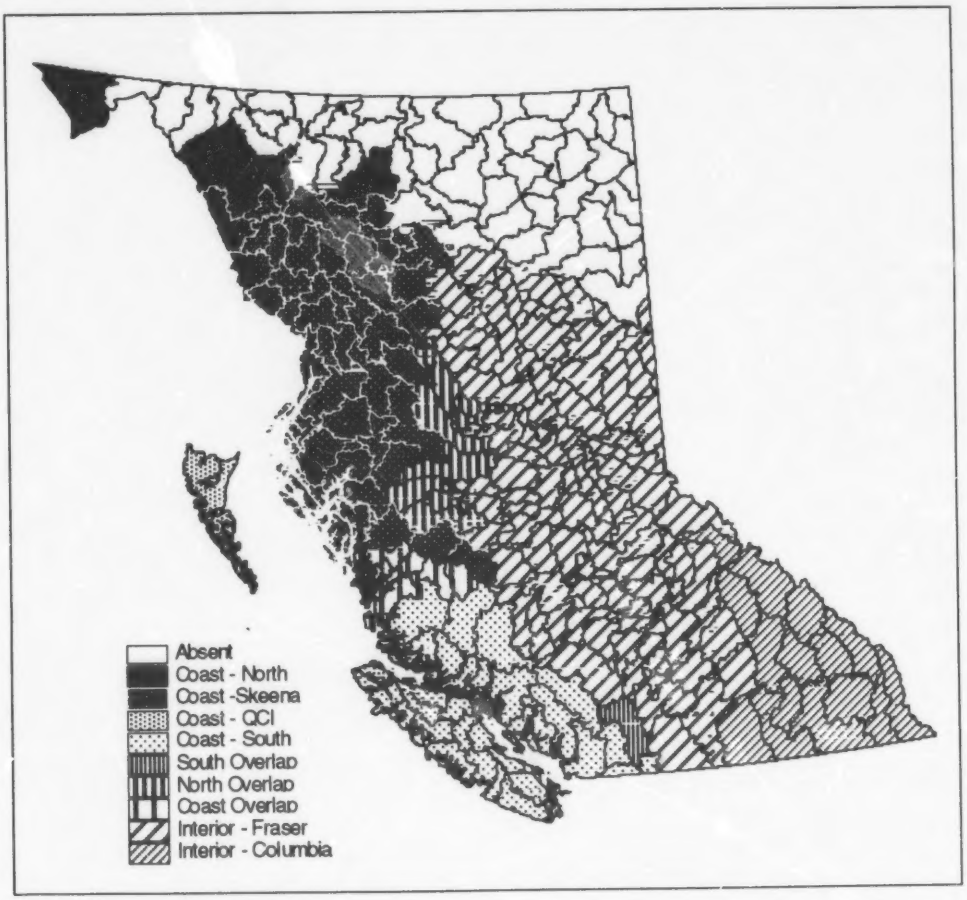

Figure 1. Current picture of rainbow/steelhead trout phylogenetic structure as inferred from molecular genetic data.

of genetic isolation can be inferred from habitat data alone. For example, molecular data from populations upstream and downstream of barriers suggest that little gene flow occurs downstream over barriers. Molecular data from steelhead suggest that there is little gene flow among adjacent watersheds. Quantitative genetic studies suggest that populations in inlet and outlet streams of lakes are often differentiated and that big lakes often contain more than 1 population of rainbow trout. Limited data suggest that there is little movement between nearby lakes but that some ecotypes associated with large lake/stream systems can have extensive migrations. These rules can be combined with the rules for species distributions to show the distribution and genetic structure of a species by map-based habitat characteristics. These data will be stored as a table in an ACCESS database which can be displayed as a layer on a GIS.

Steelhead on Vancouver Island provide a simple example of how these rules can be applied. Isozyme data (Parkinson 1984) suggest that steelhead stocks in adjacent streams are genetically distinct. In addition, ecological data (Hartman and Gill 1968) suggest that steelhead do not occupy very small watersheds. A simple rule that defines an independent population for every stream above magnitude 7 that enters the ocean directly produces a map of probable locations of all steelhead stocks on Vancouver Island (Fig. 2)

Defining Common Adaptive Phenotypes (Ecotypes)

Attempts to objectively characterize diversity or uniqueness of populations within species usually involve molecular genetic studies using polymorphic allozymes (e.g., Allendorf and Leary 1988, Leary et al. 1993); mitochondrial DNA (e.g., Perkins et al. 1993); or micro-satellite loci (Angers et al. 1995). Most molecular traits seem to be selectively neutral and are therefore ideal for defining monophyletic groups and populations that result from current or past geographic isolation. However, evolutionary significance depends strongly on adaptive traits and therefore, for conservation, the populations that share common adaptive traits must be identified. An important corollary is that successful dispersal or reintroduction to specific habitats is much more likely for populations that share common adaptive traits. Because molecular variation is not strongly coupled to the adaptive differentiation (e.g., Clayton 1981), the pattern of molecular variation is a poor tool for mapping patterns of adaptive differentiation. Others have recognized the importance of evaluating both molecular and adaptive variation in assessing differentiation among populations (MacLean and Evans 1981, Utter 1981, Waples 1991).

Unfortunately, most adaptive traits have an environmental component with inheritance controlled quantitatively by multiple genes. Many populations cannot be surveyed genetically because laborious breeding experiments are required to measure the genetic component of variation in each trait in each population. A convenient alternative is to simply use raw phenotypic variation as a surrogate for underlying genetic variation. Adaptive phenotypic traits, such as body shape or migration timing, are obvious candidates, especially if a genetic component is in the variation. However, if the goal is to classify all populations of a wide-ranging species, this simplification may not be enough. Data on habitats and biological communities, especially for characteristics (e.g., biogeoclimatic zone, stream gradient) that can be estimated remotely, offer more surrogate information for classifying populations into adaptive groups.

This approach is difficult to defend and will lead to a classification system that is prone to error. Variation in morphology, ecology, behaviour, or life history strategies typically have both genetic and environmental components (Kornfield et al. 1982, Vrijenhoek et al. 1987, Carvahlo 1993). Variation among populations in some characteristics (e.g., growth, size at maturity) are obviously dominated by environmental effects. Most conservation strategies explicitly restrict their emphasis and priorities to the conservation of genetic diversity, and implicitly or explicitly discount diversity that results from developmental plasticity (e.g., Meffe and Vrijenhoek 1988). Habitat similarity is unlikely to map on adaptive similarity if some habitat characteristics that exert powerful selective forces are omitted. Therefore, is a dependence on habitat differences as a surrogate for genetic differences justified?

Sometimes this dependence is the case. Length of freshwater migration in Atlantic salmon is associated with a number of differences among stocks that appear to be adaptations to more rigorous migration conditions (Shaffer and Elson 1975). The longer fins and fusiform bodies in Atlantic salmon parr in colder, higher gradient streams have a genetic basis and are probably adaptations to local environmental conditions (Riddell and Leggett 1981). A stock of cutthroat trout from a stream was more surface oriented in its feeding behaviour than a lake stock of cutthroat trout, when compared in a common environment (Trojnar and Behnke 1974). More generally, most phenotypic characters seem to have a genetic component and, sometimes (e.g., spawning migration time of steelhead) different populations are in a common environment when the trait is expressed. However, for most habitat variables in most species, the link between differences in habitat and differences in adaptive characteristics has not been verified quantitatively.

The immediate need for a comprehensive classification system has led us to adopt a pragmatic approach which will produce what we believe to be the most accurate picture given our current knowledge. We expect and will encourage revisions which more accurately reflect observed patterns. Each of the 3 components will be tested for its critical assumptions and refinement of the rules. Population structure rules, for example, will be tested by comparing modeling results with actual population structure as inferred from molecular data.

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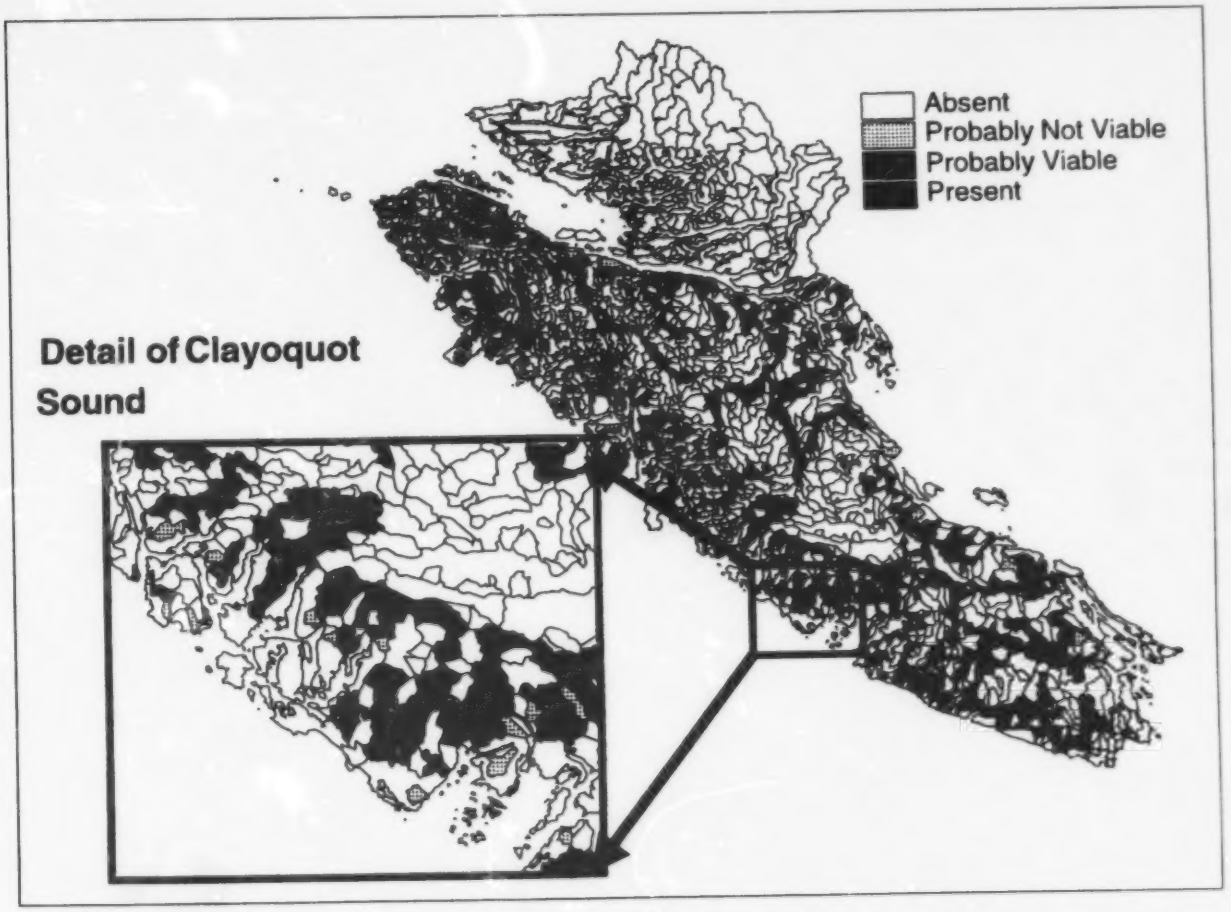

Figure 2. Model of steelhead population structure on Vancouver Island using the Watershed Atlas. The rule that generates this structure is that the stream enters the ocean directly and has a magnitude <6 (probably unviable) or >6 (probably viable) with steelhead observed (present).

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Components of a Mountain Caribou Conservation Strategy for British Columbia

Ian W. Hatter

Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9374, STN PROV GOVT, Victoria, BC, V8W 9M4, Canada
Ian.Hatter@gems6.gov.bc.ca

ABSTRACT

Mountain caribou (*Rangifer tarandus caribou*, mountain/arboreal-lichen feeding ecotype) are currently Blue-listed in British Columbia. British Columbia currently has an estimated 2,450 mountain caribou, representing about 98% of the world population, distributed in 13 spatially disjunct subpopulations, which comprise a single meta-population. Habitat fragmentation, reduction in winter food supply, human access and disturbance (including hunting and poaching), and predation have probably contributed to past declines in distribution and abundance. To meet national and international responsibilities for the long-term viability of mountain caribou, a provincial Mountain Caribou Conservation Strategy (MCCS) is being developed. The proposed vision is to "maintain caribou and their habitat in perpetuity throughout British Columbia's mountain caribou range" and the proposed goals are to: (1) maintain a population of ≥2,500 mountain caribou, distributed throughout their current range in British Columbia; (2) enhance the recovery of threatened subpopulations; (3) provide opportunities for integrated use of caribou habitats, and assess socioeconomic impacts; and (4) provide opportunities for the use and enjoyment of mountain caribou. A proposed conservation approach is outlined. Long-term conservation of the mountain caribou meta-population will require maintenance of caribou core old-growth habitats and protection of habitat linkages between subpopulations. Short-term persistence of several threatened subpopulations may require additional population management measures. Successful implementation of the MCCS will require active participation of the provincial ministries of Environment and Forests, the forest industry, and other stakeholders.

Key words: caribou, forestry, mountain caribou ecotype, *Rangifer tarandus caribou*.

CONSERVATION STATUS

All caribou in British Columbia belong to the woodland subspecies (*Rangifer tarandus caribou*), but they can be further divided into 3 different ecotypes, based on differences in habitat use, feeding behaviour, and migration patterns. Historically, the mountain/arboreal lichen-feeding caribou ecotype was more widely distributed throughout the mountainous region of southeastern British Columbia (Fig. 1). Habitat fragmentation, reduction in winter food supply, human access and disturbance (including hunting and poaching), and predation have probably contributed to past declines in caribou distribution and abundance (Stevenson and Hatler 1985, Seip and Cichowski 1996, Heard and Vagt 1998).

As far back as 1984, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) considered mountain caribou in British Columbia to be Rare, and those in the South Selkirk Mountains to be Endangered. Currently, all western caribou populations in Canada are listed as Vulnerable by COSEWIC. The trans-boundary population of the South Selkirks in the United States has been officially designated as endangered in the United States. British Columbia is currently an active member of the United States-Canada Woodland Caribou Recovery Team to recover this population (U.S. Fish and Wildlife Service 1993). Thus, British Columbia has national and international responsibilities for the long-term viability of mountain caribou.

Mountain caribou have been provincially designated as Blue-listed by the British Columbia Conservation Data Centre. As a Blue-listed ecotype, these caribou are considered vulnerable or sensitive (at risk), and need special management to ensure their survival. British Columbia currently has an estimated 2,450 mountain caribou (Table 1), which represents approximately 98% of the world's population of this ecotype. Currently, they exist in 13 spatially disjunct subpopulations, which comprise a single meta-population (Fig. 1). Some subpopulations are adjacent to each other or are connected through habitat corridors, while others are relatively isolated. At least 3 subpopulations (South Selkirk,

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Table 1. Population size, trend, and density of mountain caribou subpopulations in 1997.

Mountain caribou subpopulation	Population size	Population trend	Range (km^2)[a]	Density (no. /1,000 km2)
North Caribou Mtns.	425	increasing	6,690	64
Wells Gray South	400	stable	11,238	36
Wells Gray North	300	stable	7,654	39
Revelstoke	400	increasing	8,560	47
Hart Ranges	450	increasing	21,970	20
Central Selkirks	215	stable	5,706	38
Barkerville	40	stable	1,509	27
Narrow Lake	50	declining	432	116
Central Rockies	50	stable	9,734	5
South Selkirks	30	declining	3,456	9
South Purcells	50	declining	6,829	7
Monashee	20	declining	1,745	11
George Mountain	20	unknown	442	45
TOTAL	2,450	stable	85,964	29

[a] Present available range.

South Purcell, and Monashee) appear to be threatened, as they exist at very low density, are relatively isolated, and are currently declining.

The Mountain Caribou -Forestry Controversy

The decline of mountain caribou, particularly within the southeastern part of the province, has been recognized in British Columbia for 2 decades or more (Simpson et al. 1994). Numerous factors have been associated with caribou declines, including wildfire, predation, hunting, poaching, winter recreation, hydroelectric development, mining and mineral exploration, forest logging, corridor development (highways, railways), and human habitation (Bergerud 1978, Heard and Vagt 1998). For more than 20 years, however, forest harvesting has been recognized as the greatest management concern in mountain caribou ranges. During the past 10 years its importance has increased significantly as logging has moved into forest types that were previously avoided, particularly high-elevation subalpine fir (*Abies lasiocarpa*) and Engelmann spruce (*Picea engelmannii*) types (ESSF), and certain cedar (*Thuja plicata*) and hemlock (*Tsuga heterophylla*) types (ICH).

Stevenson et al. (1994:1) summarized the controversy as follows: "The habitat requirements of mountain caribou, as they are understood today, are incompatible with traditional forest management practices at the stand and landscape levels. Caribou are adapted to use large areas of suitable habitat. Suitable winter habitat for mountain caribou has attributes characteristic of old forests (at least 150 years), including abundant arboreal lichens (*Bryoria* spp. and *Alectoria sarmentosa*). Forests managed under any silvicultural system that eventually eliminates, or substantially reduces, the number of large, old, lichen-bearing trees will not provide winter habitat for caribou. Such silvicultural systems include clear-cut systems on normal rotations and selection systems with heavy, frequent stand entries. The problem of habitat modification by logging is compounded by associated road access, which leads to increased distur-

Figure 1. Present and potential historic distribution of mountain caribou. The present distribution includes 13 subpopulations.

hance, human-induced mortality, and potentially to increased predation by wolves (and cougars)."

PROPOSED VISION AND GOALS

The proposed vision for the provincial Mountain Caribou Conservation Strategy (MCCS) is "the maintenance of caribou and their habitat in perpetuity throughout British Columbia's mountain caribou range." This vision reflects the social, cultural, and economic values associated with mountain caribou—including people and caribou living in harmony. The proposed goals of the strategy are:

1) *To maintain a stable population level of ≥2,500 mountain caribou distributed throughout their current range in British Columbia.*— This is the overriding goal for mountain caribou conservation. It seeks to ensure that a viable meta-population is maintained, and that mountain caribou remain an integral component of the large mammal fauna within the Southern Interior Mountains ecoprovince and the Hart Ranges ecosection. While it would be preferable to increase mountain caribou numbers to a level where they could be removed from the Blue List, this may not be achievable, given the need to simultaneously address goal 3.
2) *To enhance the recovery of small, threatened subpopulations.*— Several subpopulations appear to be threatened and could become extirpated unless specific recovery measures are implemented. While the long-term conservation value of these subpopulations for maintaining a viable meta-population is presently unknown, it may be significant. Wherever possible, approved recovery plans should be developed for threatened subpopulations. This goal also recognizes British Columbia's continued commitment to recovery of the "endangered" mountain caribou subpopulation within the South Selkirk ecosystem (U.S. Fish and Wildlife Service 1993).
3) *To provide opportunities for integrated use of caribou habitats and assess socio-economic impacts.*— This goal recognizes that maintaining or enhancing mountain caribou habitat has socioeconomic impacts on other land uses and local communities. Opportunities to provide for integrated use of caribou habitat must be sought, providing they are consistent with mountain caribou conservation. The most effective means for satisfactorily resolving human conflicts with mountain caribou is through research and adaptive management, and by ensuring full participation of all relevant stakeholders in the decision-making process. Thus, responding to the challenge of integrating resource use and caribou management will be a long-term process, based upon negotiation and compromise. That being so, negotiation should be based upon a provincial assessment of conservation risks and economic impacts (Simpson et al. 1997). These include:
   - a comparative evaluation of conservation values of the 13 subpopulations;
   - a comparative evaluation of the economic costs of maintaining the 13 subpopulations;
   - an assessment of the variety of integrated management options available; and
   - an adequate knowledge of mountain caribou ecology.

   The appropriate forum for assessing socioeconomic impacts of protecting caribou habitat will likely be through the regional and subregional land-use planning processes (Land and Resource Management Plans [LRMPs] and the Forest Practices Code's Higher Level Planning processes). Land-use decisions that have already been made by cabinet must also be recognized and accepted.
4) *To provide opportunities for the use and enjoyment of mountain caribou.*— Part of British Columbia's wildlife heritage is the use and enjoyment of wildlife by people. The priority use of mountain caribou is conservation; that is, maintain subpopulations at self-sustaining levels and translocate (transplant) individuals to recover other subpopulations. Other approved uses, where compatible with the overriding goal of conservation, include viewing, First Nations use, and resident/non-resident hunting.

Conservation Approach

Management of mountain caribou must weigh the risks or threats of various management actions with conservation needs. To address this, the MCCS should develop a conservation approach that recognizes the meta-population structure of mountain caribou, employs the precautionary principle, encourages learning through adaptive management, and is based on ecosystem management principles.

Meta-population Persistence

A successful strategy for mountain caribou must recognize their meta-population structure. Dispersal from larger subpopulations augments and possibly even "rescues" small populations (prevents extirpation and ensures genetic interchange for adaptability). Conversely, small subpopulations could be important for recovering larger subpopulations that suffer a catastrophic event. Maintaining subpopulations, as well as "habitat linkages" or dispersal routes between them, is critical to meta-population persistence. Small, isolated populations, which have very little chance for dispersal, may require periodic herd augmentation (transplants) to prevent extirpation and maintain genetic variation.

Precautionary Principle

Application of the precautionary principle to mountain caribou means that where there are threats of serious habitat loss associated with forest development or other industrial activities, lack of full scientific certainty should not be used

as a reason for altering caribou habitats. This principle places the onus on both government and industry (e.g., through joint ventures such as Forest Renewal British Columbia) to demonstrate that environmental impacts that alter mountain caribou habitat, habitat use patterns, or demography, have either nil, negligible, or an acceptable level of impact.

Adaptive Management

While conventional research studies have greatly improved our knowledge of mountain caribou and their habitat, an adaptive management approach is necessary to provide long-term solutions on how to mitigate the impacts of timber harvesting on caribou. However, because the response time of mountain caribou habitats to alterations and management experiments is long, results can be slow in appearing (Armleder and Stevenson 1996). Consequently, interim habitat management guidelines based on existing research, the precautionary principle, and ecosystem management principles should be developed until more definitive results from adaptive management experiments are available.

Ecosystem Management

Ecosystem management is based on the assumption that the more closely managed forests resemble natural forest conditions (i.e., age class distribution, patch size distribution, stand structure), the greater the probability that relatively natural populations of all native species will be maintained (Seip 1998). The MCCS should consider employing a comprehensive ecosystem management approach for mountain caribou. For example, when dealing with ESSF zone forest management issues, it would also be appropriate to stress the multiple conservation values of that zone, both to mountain caribou and to other species, including Blue-listed fish like bull trout (*Salvelinus confluentus*), as well as the production of water supplies for aquatic ecosystems and industrial and domestic use.

MANAGEMENT PRESCRIPTIONS

Current knowledge suggests that the long-term persistence of mountain caribou is dependent upon a perpetual supply of large, contiguous areas of suitable summer and winter habitat, with little or no vehicle access and human disturbance, so caribou can space out at low densities (approximately 40/1000 km2), and thus reduce predation risk (Seip and Cichowski 1996). However, most subpopulations occupy ranges that have been fragmented to varying degrees by logging and access, and have higher predator populations than occurred historically, due to alteration of predator-prey systems. Thus, a combination of both habitat and population management prescriptions will likely be required to maintain viable caribou subpopulations.

Habitat Management

Some form of caribou habitat management guidelines or planning/operational direction is in place throughout mountain caribou range. The impact of these guidelines on the forest industry vary, depending on the extent of conflict between caribou and timber, and the differing regional behavioural and movement patterns of caribou. Current habitat management efforts are focused on the following 4 major issues:

Identify and Protect Core Habitat Areas

At 1 time, most herds included a high-elevation no-harvest zone, which in most areas corresponded approximately to the forest harvesting "operability line" (Simpson et al. 1997). More recently, the Cariboo-Chilcotin Land Use Plan (CCLUP) has replaced a 20-year deferral of upper-elevation, late-winter habitats with a zone that allows up to 35% modified timber harvesting for each CCLUP subunit. Other LRMPs have attempted to maintain 30–40% of the operable landbase in age class 8 or older (>160 yr) within high-elevation, late-winter habitats.

Establish Alternative Silvicultural Systems

The primary focus of the Mountain Caribou in Managed Forests program was to identify silvicultural systems and habitat enhancement techniques that would sustain both timber harvest and mountain caribou winter habitat over the long term (Stevenson et al. 1994). Many of the adaptive management trials established under that program are still underway and require additional testing before implementation.

Maintain Existing Corridor or "Linkage" Areas

Travel corridors of snow-interception habitat should be maintained between low and high elevations to enable caribou movement between seasonal ranges. Current connectivity between subpopulations also needs to be maintained by retaining existing corridors of suitable caribou habitat within identified linkage routes.

Access Management

All subpopulations should require an access management plan to ensure that recreational activities within mountain caribou range, such as use of all-terrain vehicles and snowmobiles, do not negatively impact mountain caribou. The preferred approach for regulating outdoor recreational activities will likely be through the regional and subregional land-use planning process.

Future work though the MCCS should focus on improving the technical/scientific basis for managing caribou habitats and facilitating coordination of regional habitat prescriptions. Maintaining caribou core old-growth habitats, establishing alternative silvicultural systems, regulating recreational activities within caribou range, and protecting habitat linkages between subpopulations should continue to be a high priority.

Population Management

Population management prescriptions have traditionally involved harvest management and herd augmentation by translocation. Our current understanding suggests that managing other ungulate species within mountain caribou habitat and predator management may be equally or even more important for long-term viability of mountain caribou.

Harvest Management

Currently, there is a moratorium on mountain caribou hunting in British Columbia until conservation concerns have been addressed. This should be maintained.

Translocation

Translocation has been used to increase mountain caribou numbers within threatened subpopulations (e.g., South Selkirk subpopulation). Its continued use as a conservation measure will be determined by the availability of source animals for transplant, and the ability to manage predators and other ungulate species within mountain caribou habitat.

Managing Other Ungulate Species within Mountain Caribou Habitat

Maintaining or lowering alternate ungulate prey densities (i.e., deer, elk, and moose) through hunting, as well as curtailing habitat enhancement activities for these species, may be necessary to keep predator densities low within areas occupied by threatened mountain caribou subpopulations.

Predator Management

Several recent studies have implicated predation as an important proximate cause of mountain caribou mortality (Simpson et al. 1997). In some circumstances, it may be necessary to increase predator hunting and trapping opportunities in order to conserve caribou. In extreme situations, where continued predation could extirpate subpopulations, it may be necessary to employ other more intensive lethal or nonlethal means to effectively reduce predation rates on caribou.

Future work though the MCCS should focus on implementing all 4 population management activities within threatened subpopulations. Due to a shortage of "source" mountain caribou for transplants, attention should be given to investigating the genetic and behavioural implications of augmenting these subpopulations with the more numerous northern/terrestrial lichen-feeding caribou ecotype from north-central British Columbia.

STRATEGY IMPLEMENTATION

Successful implementation of the MCCS will require active participation and cooperation between the Ministry of Environment, the Ministry of Forests, the forest industry, and other non-government stakeholders. A Mountain Caribou Technical Advisory Committee, with representation from these organizations, has recently been established to develop a provincial MCCS. This committee has also been charged with developing a list of conservation priorities and activities that will help to ensure the vision and goals for mountain caribou are achieved.

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ADDENDUM (MAY 2000)

Mountain caribou are now Red-listed by the Conservation Data Centre due to current threats and changes in herd status. The current estimate is 2300 caribou. Five herds are declining, 8 herds are stable and none are increasing.

Connectivity and the Movements of Radio-Collared Caribou

Douglas C. Heard
British Columbia Ministry of Environment Lands and Parks
Prince George, BC, V2L 3H9, Canada
Doug.Heard@gems7.gov.bc.ca

Glen S. Watts
British Columbia Ministry of Environment Lands and Parks
Prince George, BC, V2L 3H9, Canada

EXPANDED ABSTRACT

Key words: connectivity, global positioning system, mountain caribou, radiotracking, *Rangifer tarandus caribou*.

Over the long term, mountain caribou (*Rangifer tarandus caribou*) conservation will almost certainly require the maintenance of connectivity among herds, but the characteristics of what constitutes connecting habitat are neither obvious nor easily determined. We felt that GPS (global positioning system) radio-collars might provide the frequency of location data required to detail caribou travel paths, and that those data might provide some indication of what might constitute caribou movement corridors.

We collared 3 adult female caribou from the Yellowhead herd (Heard and Vagt 1998) east of Prince George, B.C. in spring 1996, and 4 caribou from the same herd in early 1997, using Lotek GPS 1000 radio-collars. A variety of technical failures reduced the number of times that the collars attempted to acquire a position from an anticipated 15,000 to only 4,000. Because only about 50% of the attempts were successful, we ended up with only 1,900 useable locations.

To distinguish travelling from foraging and other movements, we used a nonlinear, curve-fitting approach on the log of the movement rate frequency data to separate activity bouts (Sibly et al. 1990). We distinguished 3 travelling (i.e., high movement rate) periods in our data set: 1 by caribou CA, and 2 by caribou CB.

CA was travelling almost the entire time her collar functioned, which was from 29 March 1996 to 2 May 1996. During that period the collar obtained 138 useable location fixes, even though it was scheduled to do so every hour. CA moved generally east to west when travelling until she came to the eastern edge of the Bowron clearcut, an opening of roughly 7,000 km^2 that was logged in about 1980. At that point she swung north, generally paralleling the clearcut boundary, but never entering or crossing, even when the opening narrowed near Tumuch Lake. She also appeared to avoid other smaller cutblocks scattered around the periphery of the Bowron clearcut and generally in her path of movement. The absence of any locations in cutblocks was not likely due to a bias in the GPS collar, since collars are more likely to be successful at obtaining a fix in the open than in the forest.

CB's collar functioned from 22 April 1997 until 29 June 1998, obtaining 627 useable location fixes, with attempts scheduled every 6 hours. The first travelling period was in late June and July, when she moved in almost a straight line across the clearcut at Tumuch Lake and reversed her direction of movement only when encountering the Yellowhead Highway. If the location data and the highway map are both precise, CB appeared to have crossed the highway before turning around and recrossing, moving back up to higher-elevation forests. She remained in the same general area until September, when her movement rate again increased substantially. At that time she descended to the highway, but did not appear to cross before turning around and returning to where she had come from and where she eventually spent the winter.

Direct observations of caribou along the highway, and movements of radio-collared caribou that were documented during a previous study, indicated that caribou had on numerous occasions moved across the Yellowhead Highway. Moreover, they appeared to have crossed in, and only in, the same area where CB went but did not continue through. None of the caribou monitored in previous years crossed the clearcut at Tumuch Lake.

Because of our small sample size and the variability between animals, we were unable to draw any conclusions about habitat connectivity characteristics from our data. CA appeared to be reluctant to cross the clearcut at Tumuch Lake at the same point where CB crossed, but CB did not cross the Yellowhead Highway on either of 2 occasions, even though many other caribou had done so previously. The large number of locations that GPS collars can provide has the potential to refine and improve our understanding of caribou habitat use, but only after a much larger number of individuals has been monitored.

ACKNOWLEDGEMENTS

We thank the Columbia Basin Fish and Wildlife Compensation Program and the British Columbia Ministry of Environment, Lands and Parks for funding this work.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Caribou, Kamloops, B.C. 520pp.

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Population Characteristics of the Lake Revelstoke Caribou

John P. Flaa
Parks Canada
P.O. Box 350, Revelstoke, BC, V0E 2S0, Canada
john_flaa@pch.gc.ca

Bruce N. McLellan
British Columbia Ministry of Forests
RPO#3, P.O. Box 9158, Revelstoke, BC, V0E 2S0, Canada

ABSTRACT

The Lake Revelstoke caribou (LRC) are the southernmost contiguous mountain caribou population in British Columbia. Population characteristics of the LRC were examined during 1981–85, with 13 radio-collared caribou, and 1992–98, with 47 radio-collared caribou. During several late-winter censuses, sightability of marked animals was 92% and the proportion seen was a function of the height of snow (HS). Our best estimate of the population size was 375 animals (95% CI 337–413). Cause of death of 31 radio-collared animals included accidents (42%), predation (29%), and poor condition (10%). Deaths due to avalanches (accidents) were the single leading cause at 23%. Survival of radio-collared caribou averaged 84% since 1992. Progesterone levels in 29 female caribou indicated that 83% were pregnant. There was an average of 28 calves/100 adults during 1981–84, and 18 calves/100 adults during 1994–97. Survival rates of adult caribou and the ratio of calves per 100 adults indicate that the population was stable.

Key words: caribou, census, mortality, pregnancy rate, *Rangifer tarandus caribou*, recruitment, survival rate.

Woodland caribou (*Rangifer tarandus caribou)* are listed as Vulnerable by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), and those in southern British Columbia are Blue-listed (vulnerable) by the British Columbia Conservation Data Centre. Woodland caribou in southern British Columbia are a unique ecotype characterized by their use of deep snow as a platform to access arboreal lichens in subalpine forests during winter (Stevenson and Hatler 1985). The Lake Revelstoke caribou (LRC) are the southernmost contiguous mountain caribou population in British Columbia. Population characteristics of the LRC were examined during 1981–85 (Simpson et al. 1987), and during 1992–98 (McLellan and Flaa 1993, McLellan et al. 1994*a*,*b*). This paper summarizes the population characteristics of the LRC during these studies.

STUDY AREA

The study area was located north of Revelstoke, B.C. (51°N, 118°W) and included the Selkirk Mountains to the east of the Revelstoke Reservoir, and the Monashee Mountains to the west of the reservoir. Elevations range from 550 to 3,050 m, with tree line at approximately 1,980 m. Lower slopes are in the wet-cool Interior Cedar–Hemlock (ICHwk) biogeoclimatic subzone. Mid-slopes are in the very wet cold Engleman Spruce–Subalpine Fir (ESSFvc) subzone. Alpine, rock, and glaciers of the Alpine Tundra (AT) zone occur above the ESSF. The area of transition between the ESSF and the AT is characterized by clumps of subalpine fir forming open subalpine parkland (Pojar et al. 1987). Timber harvesting and hydroelectric development are major industrial uses of the area and have changed the landscape and forest cover of the study area.

METHODS

Between 1981 and 1985, 13 caribou were captured by darting with Anectine (succinylcholine chloride) from the ground, and in 1992–98, 47 caribou were captured by net-gunning from a helicopter in subalpine habitat. All caribou were fitted with mortality/motion-sensitive radio-collars. Aerial relocation schedules for both study periods were bi-weekly, with increased coverage in early winter. Blood samples were collected from 29 female caribou during the 1992–98 study and analyzed for progesterone concentration to determine pregnancy.

When mortality signals were detected during monitoring flights, the site was investigated as soon as possible. Caribou were necropsied, with carcass position, evidence of bleeding, predator sign, and bone marrow fat content used to aid in determining contributing factors and primary cause of death. Survival rates were determined using BOOTER (Hovey and McLellan 1996), while MICROMORT (Heisy and Fuller 1985) was used to determine seasonal differences in mortality rate.

Caribou were censused during late winter, when they were

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

in open subalpine parkland (Simpson et al. 1987, Seip 1992*b*, McLellan et al. 1995). Censuses in 1994, 1996, and 1997 were conducted shortly after new snowfalls in 3 discrete census units from a Bell 206 helicopter with 3 observers plus the pilot. A contour near timberline (1,800–2,130 m) was flown, including all suitable habitat within the study area, and tracks were followed until animals were sighted, unless the tracks descended into mature timber and were lost from view. Collars were used as marks and not to locate animals. Classification was limited to adults or calves in mixed groups and males in segregated all-male groups.

An earlier census in 1985 by Simpson et al. (1987) was conducted using a similar methodology.

Chapman's (1951) formula was used to estimate the population size:

$$N = [(M + 1)(C + 1)/(R + 1)] - 1$$

where N = population size, M = the total number of caribou marked, C = the total number of caribou seen during census, and R = the number of these marked. Confidence intervals were determined by Poisson distribution (Krebs 1989). Growth rate (λ) was determined by:

$$\lambda = (1 - M)/(1 - R)$$

where M = adult mortality rate and R = calf recruitment rate (Bergerud and Elliot 1986).

RESULTS

Plasma progesterone levels indicated that 24 of 29 female caribou (83%) were pregnant. Progesterone levels for pregnant caribou varied from 8.0 to 25.0 nmol/l, and were <2 nmol/l for nonpregnant caribou (Ringberg and Aakvaag 1982, Seip 1990, Seip 1992*b*; Fig. 1).

Deaths of 31 adult radio-collared caribou were recorded. Body-condition-related deaths, and accidents accounted for 52% of the mortality. Predation accounted for 29% of the deaths, and 19% died from unknown causes (Table 1).

There was no difference in mortality among seasons when adjusted by season length. Seasonal survival rates were between 0.93 and 0.97, with all 95% CI overlapping (Table 2).

Based on 26 mortalities from 1992 to 1998, with 177 caribou-monitoring years, adult annual survivorship varied between 71% and 92% over individual years. Cumulative annual survivorship was 84% (95% CI 78–89%). The 1981–85 survival rate was 81%, with 5 mortalities in 21 caribou-monitoring years (Simpson et al. 1987).

Estimates of the number of caribou in the LRC area are shown in Figure 2. Due to variable census methods, survey intensities, and winter conditions, any suggested trend should be viewed with caution.

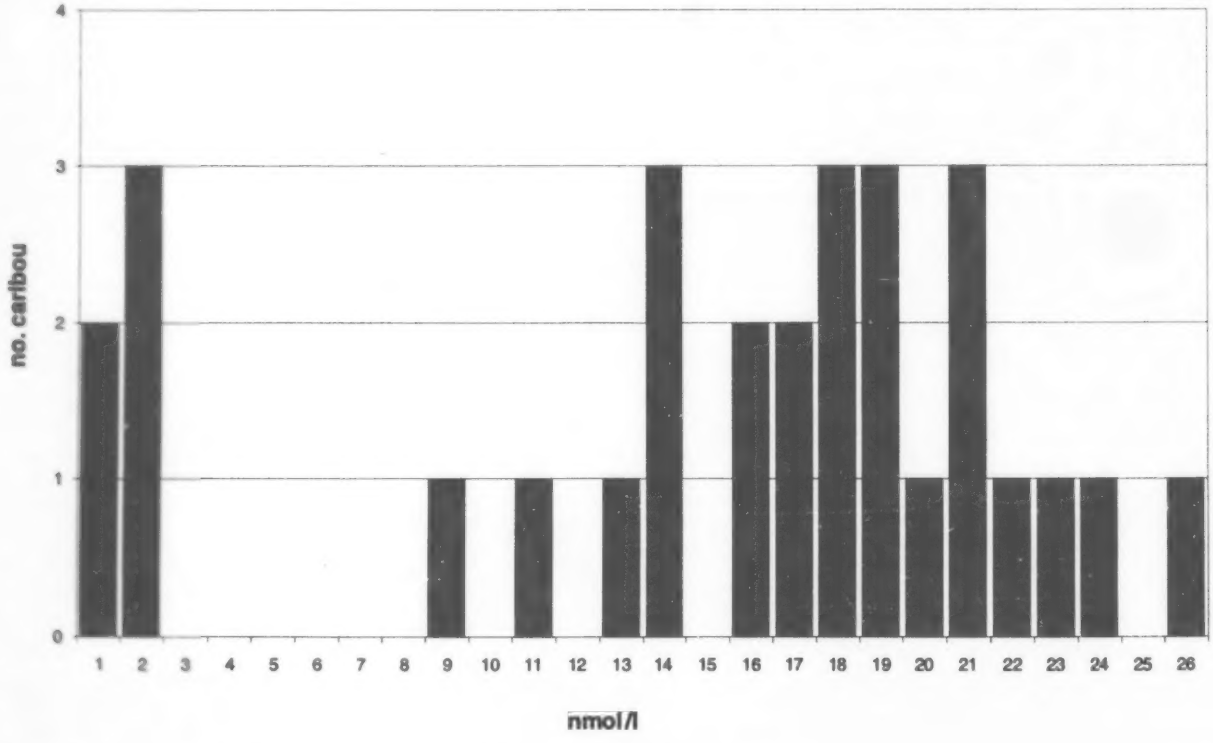

Figure 1. Progesterone concentration in 29 female caribou captured in March, 1992–1996.

Table 1. Mortality causes of radio-collared caribou in the Lake Revelstoke study area, 1981–85 and 1992–98. Percentages of total caribou mortalities are in parentheses.

Cause of death	No. of caribou (% of total mortalities)
PREDATION	9 (29)
Bear	3 (10)
Cougar	4 (13)
Wolverine	2 (6)
NONPREDATION	16 (52)
Avalanche	7 (23)
Accidental	6 (19)
Condition	3 (10)
UNKNOWN	6 (19)

Since 1993, 3 censuses (1994, 1996, 1997) were completed and 1 census (1993) was aborted due to low sightability of marked animals. During the 1994, 1996, and 1997 censuses, 347, 268, and 308 caribou were observed respectively. Population estimates of caribou were 375, 290, and 339, with upper 95% CI of 590, 439, and 496 (Fig. 2).

During the censuses we saw 24 of 26 marked animals in 1994, 23 of 25 in 1996, and 26 of 28 in 1997, for an average sightability of 92%. There was a significant relationship between height of snow (HS) at 1,830 m elevation and the sightability of marked animals ($r2 = 0.93$, $p = 0.002$; Fig. 3).

During 1981–84, an average of 22% of the population in March was calves. The average for 1994, 1996, and 1997 was 15% calves, with a range 12–17%.

Population growth rate was estimated to be 1.01 for 1981–85, and 0.98 for 1994–97.

Table 2. Estimated seasonal survivorship of Lake Revelstoke caribou, 1981–85 and 1992–98.

Season	Survival rate	Upper 95% CI	Lower 95% CI
Early winter	0.973	0.999	0.948
Late winter	0.958	0.991	0.926
Spring	0.966	0.996	0.937
Summer	0.933	0.972	0.896
Annual	0.841	0.898	0.787

DISCUSSION

The pregnancy rate found in the LRC is similar to the 76–100% recorded for woodland caribou in other areas (Seip 1992*a*, Crete et al. 1996, Seip and Cichowski 1996). There was a high loss of calves from the population in the first year of life. During March we found an average of 18 calves/100 adults, which indicates a loss of 23 calves/100 adults if we assume adult males and females were in equal proportions in the population, which is unlikely true, and thus even more calves were lost. This high loss of calves is similar to that reported for other populations of woodland caribou in British Columbia (Bergerud and Elliot 1986; Seip 1992*a*,*b*; Seip and Cichowski 1996).

Death of adult caribou due to accidents was common in this study but was not reported for other woodland caribou populations. This high accident rate was attributable to the rugged topography and extreme snow accumulation. Wolves, which occur but are not abundant in the study area (Allison 1998), did not kill radio-collared caribou. Bear predation occurred during summer, and wolverines preyed on weak or injured caribou in the winter. The incidence of cougar predation appears to have increased during the study, with 3

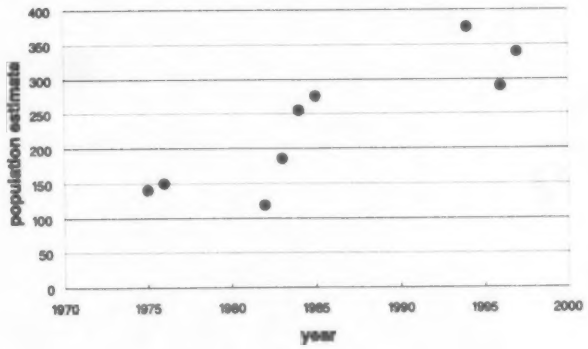

Figure 2. Population estimates of the Lake Revelstoke caribou, 1975–1997.

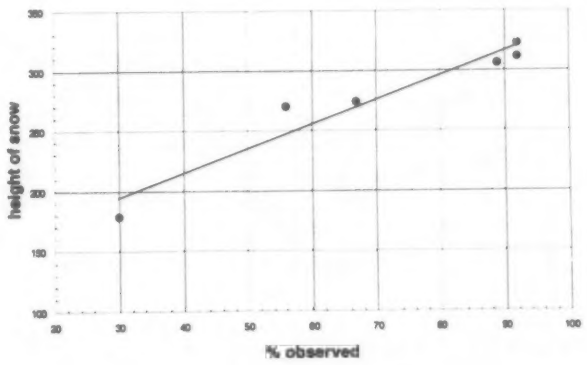

Figure 3. Percentage of radio-collared caribou observed during censuses relative to height of snow (HS) at 1,830 m on Mount Revelstoke.

of the 4 incidents occurring since 1996 and none reported by Simpson et al. (1987).

Unlike other studies in British Columbia, where the major period of mortality occurred between May and October (Seip and Cichowski 1996), mortality in our study showed no significant difference among seasons. This balance of seasonal mortality was due to the increased winter-related accidental mortality of the LRC, compared to other studies where predation during summer and fall was a major cause of death (Bergerud and Elliot 1986; Seip 1992*a*,*b*; Rettie et al. 1998).

Census results indicate that there were 350–400 caribou in 1997. Simpson et al. (1987) estimated the population to be 275–300 in 1985; however, that estimate may be low, as their coverage of the study area was not as extensive as in 1994, 1996, and 1997.

Major limiting factors include high calf mortality and accidental adult mortality due to the rugged terrain and extreme winter snowfalls. Increased cougar predation appears to have become a more significant factor and may require management action. Loss of habitat is not thought to be a major limiting factor at present; however, it will likely become more significant as the forest profile changes from older stands and young plantations to a forest fragmented by younger stands.

ACKNOWLEDGEMENTS

We thank D. Mair for his skill and dedication flying in often difficult conditions in the wet-belt ecosystem, and J. Hooge and M. Super for locating and recording radio-collared animals. F. Hovey assisted with statistical analysis. We also appreciate those who assisted with caribou capture. Funding was provided by Parks Canada, the British Columbia Ministry of Forests, the Columbia Basin Fish and Wildlife Compensation Program, and Forest Renewal British Columbia.

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Recent Augmentation of Mountain Caribou to the Southern Selkirk Mountains of Northeastern Washington, Northern Idaho, and Southern British Columbia

J.A. Almack

Washington Department of Fish and Wildlife
12645 Sullivan Lake Rd., Metaline Falls, WA 99153-9701 USA
almackfw@iomet.com

ABSTRACT

Over a 3-year period, we transplanted 43 radio-marked mountain caribou from central British Columbia to the southern Selkirk Mountains of Washington, Idaho, and British Columbia. We have monitored these animals to determine initial movements from release site, seasonal movements, habitat use, recruitment, and cause of mortality. Five caribou have emigrated from the Selkirk Mountains recovery zone; 2 returned to the area, 3 died within weeks of leaving. Other movements and habitat use are still being analyzed and will be completed in time for presentation at the conference. We have observed calf production each year, but survival of this cohort is unknown. Twenty-six radio-collared caribou have died: 4 were killed by cougars, 1 by grizzly bears, 1 from accidental fall, 2 from poaching, and 18 from unknown causes. Monitoring of this population will likely continue for at least 2 more years. We also have initiated a cougar study in cooperation with the United States Fish and Wildlife Service, United States Forest Service, and British Columbia Wildlife Branch to determine the effects of predation on the survival of this endangered caribou population.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 529pp.

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A Management Strategy for Mountain Caribou: The Cariboo Region Example

Harold M. Armleder
British Columbia Ministry of Forests
200-640 Borland Street, Williams Lake, BC, V2G 4T1, Canada
Harold.Armleder@gems6.gov.bc.ca

James A. Young
British Columbia Ministry of Forests
200-640 Borland Street, Williams Lake, BC, V2G 4T1, Canada

John A. Youds
British Columbia Ministry of Forests
200-640 Borland Street, Williams Lake, BC, V2G 4T1, Canada

ABSTRACT

We describe how research and monitoring were applied, within higher level plan direction, to craft an integrated management strategy for mountain caribou. Nine years of radiotelemetry on 75 animals with >4,000 relocations have defined the range and habitat selection patterns for this population. Timber harvesting trials since 1990 have tested variations of the selection silvicultural system. The Cariboo-Chilcotin Land-Use Plan has established the size of the area that can be managed for mountain caribou and defined the level of acceptable impact on the timber resource. These inputs were used to develop an initial strategy which zones caribou habitat and describes management in those zones. Recommendations for forest management systems, access, and predation management are part of the strategy.

Key words: access management, land-use planning, logging, mountain caribou, predation, *Rangifer tarandus caribou*, silvicultural systems.

The mountain caribou (*Rangifer tarandus caribou*) within the Cariboo Forest Region (Wells Gray North, Barkerville, and a portion of the North Cariboo Mountains subpopulations) comprise about 300 animals and represent 12% of the provincial mountain caribou population. The importance of these caribou has been provincially recognized (Simpson et al. 1997). In 1980, the British Columbia Ministry of Environment, Lands and Parks delineated high elevation caribou habitat and subsequently negotiated a 20-year timber harvesting deferral over much of the original area of concern. The mid-1980s brought an increase in population surveys and the first radiotelemetry research which established basic habitat use patterns (Seip 1992*a*).

Mountain caribou require old forest habitat to obtain the arboreal lichen they need for winter food (Stevenson et al. 1994). Clearcutting removes the entire arboreal lichen food resource whereas the new forest will be 100–150 years old before arboreal lichens are abundant enough to be suitable forage for caribou (Armleder and Stevenson 1994). Consequently, researchers at the British Columbia Ministry of Forests began to investigate integrated timber management solutions in 1989. In 1990, a pilot block was harvested using several partial cutting treatments. Because arboreal lichen is a vital winter food, lichen growth rates and biomass changes in various treatments were also measured. In 1992–93, a major multidisciplinary, replicated silvicultural systems trial was established.

In 1993, the Ministry of Environment, Lands and Parks initiated an intensive radiotelemetry project continuously involving 20–25 caribou relocated 2–4 times/month (>4,000 relocations as of January 1999). In addition, wolves from 3 packs were monitored using radiotelemetry. Trailing research was also conducted to explore caribou habitat use at the stand level. To meet the anticipated data needs arising from the end of the timber harvesting deferral, a major terrestrial ecosystem mapping project covering all the mountain caribou range was completed.

CARIBOO-CHILCOTIN LAND-USE PLAN

While the research was ongoing, regional land use planning was initiated with a plan in 1995. The Cariboo-Chilcotin Land-Use Plan (CCLUP) recognized that mountain caribou in the eastern part of the region are provincially significant and are a species at risk. Maintaining habitat values for mountain caribou was identified as an overriding objective within the plan. To address this objective, the CCLUP mandated a caribou strategy that would recommend how habitat and other factors impacting caribou could be managed. An initial strategy was produced in 1996 with an update in 1998. The final strategy is due in 2000. The CCLUP recognized the value of

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

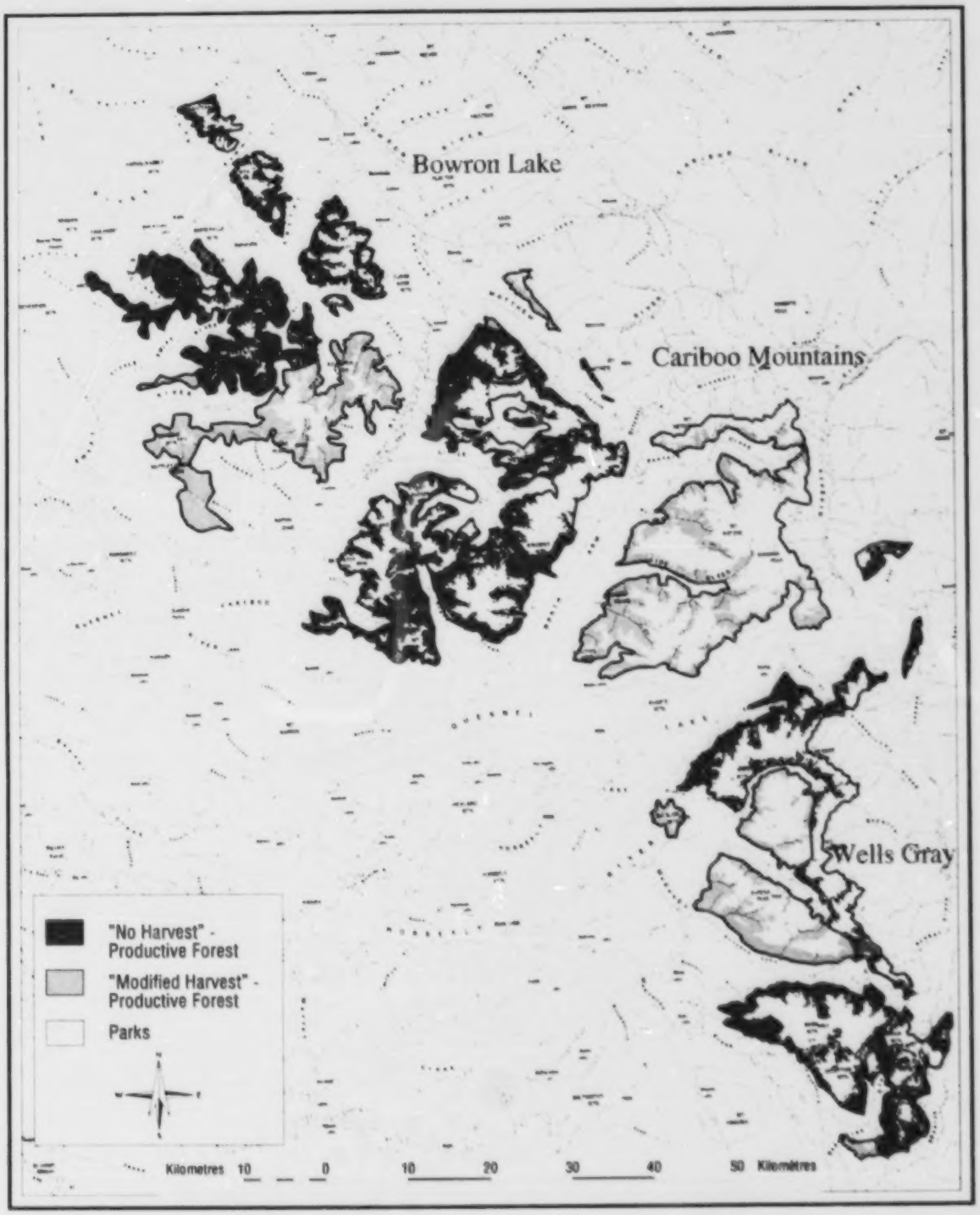

Figure 1. Zonation of mountain caribou habitat within the deferral area in the Cariboo Forest Region, British Columbia, 1998. Of the productive forest, 35% is zoned "modified harvest" whereas 65% is zoned "no harvest" within the deferral area.

ongoing research and therefore deferred the full implementation of the plan for caribou habitat until the year 2000. The entire caribou strategy process involved consultations with the forest industry and the conservation council as well as government agencies. This integrated management approach has led to smoother strategy implementation.

Our paper will review the current strategy (Caribou Strategy Committee 1998) and reference the role played by research, monitoring, and higher level direction to the development of the strategy.

ZONING OF HABITAT

Input from the CCLUP resulted in the adoption of zoning the caribou habitat. The CCLUP specified that 65% of the productive forest land within the deferral area would be zoned "no harvest" and the remaining 35% as "modified harvest" (British Columbia 1995). The Caribou Strategy Committee used the following criteria to delineate the location of "modified" harvest areas for mountain caribou (Fig. 1):

1. Areas of lower caribou use (using data from radiotelemetry studies).
2. Areas of lower suitability for caribou (from terrestrial ecosystem mapping).
3. Areas of lower human accessibility because development will create access concerns.
4. Large areas (typically thousands of hectares) to complement large "no harvest" areas. Additionally, large areas offer a full range of elevation, slope, aspect, and timber types for forest companies to access.
5. Areas peripheral to the range and therefore of less value to caribou.

Interestingly, the deferral drawn in 1980 accurately identified high elevation caribou habitat when considering the >4,000 radiotelemetry relocations. Exceptions occurred mostly in the Barkerville subpopulation and are being addressed through possible trades of less valuable areas within the deferral for more recently identified habitat areas outside the deferral. The overriding constraint is that the impact to timber extraction can be no greater than that modelled in the CCLUP. This constraint limits the protection of low elevation early winter range in the Interior Cedar–Hemlock (ICH) zone.

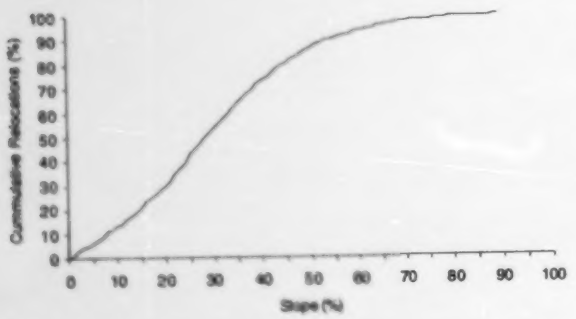

Figure 2. Cumulative frequency distribution of slope use by radiotelemetry-equipped mountain caribou for all seasons combined in the Cariboo Forest Region, British Columbia, 1993–1998.

TIMBER HARVESTING STRATEGY

All recommended approaches were designed to be fully compatible with the impacts to timber modelled in the CCLUP (Integration Committee 1998). A 3-fold approach for managing the habitat of mountain caribou is recommended.

1. Park and "No Harvest" Areas

These areas provide a core habitat that will have little or no road access (very limited salvage harvesting in "no harvest" areas is permitted according to the CCLUP Integration Report: Integration Committee 1998). This approach provides caribou not only with suitable space (habitat) in which to meet their needs but also a large area free of harvest and harassment by humans, provided the use of snowmobiles and all-terrain vehicles (ATVs), and other activities are adequately controlled. Because logging will not create more early seral habitat (or very little as in the CCLUP Integration Report), moose will not be enhanced which will help protect caribou from additional predation pressures caused by increased wolf numbers. These areas provide a perpetual supply of large, contiguous suitable habitat so that caribou can space out at low densities and avoid predators and harassment (Seip and Cichowski 1996).

2. "Modified Harvest" Areas

These areas will be managed to maintain caribou habitat continuously through time and space and will emulate natural disturbance patterns. Harvesting approaches are designed to discourage the enhancement of moose habitat.

"Modified Harvesting" Approach on Arboreal Lichen Sites <45% Slope

Mature and old stands with a slope <45% are most heavily used by caribou (Fig. 2) and therefore should be managed to maintain habitat value continuously through time and space. This strategy should be accomplished as follows:

Maximize the size of cutblocks as allowed in this natural disturbance type (NDT) in the Forest Practices Code (FPC) *Biodiversity Guidebook* (B.C. Ministry of Forests, and B.C. Ministry of Environment, Lands and Parks 1995). This will serve to spatially concentrate harvesting activity.

Harvest by applying 33% volume removal group selection with an 80-year cutting cycle. This approach is supported by 8 years of experimental research which, although not

completed, shows encouraging preliminary results for forest management and for caribou.

Harvest openings should be 2–3 tree-lengths wide and up to 3–4 tree-lengths long. Where windthrow and wind-scouring of lichens are not concerns the openings could be somewhat longer. Openings should be <1 ha in size.

This management approach is not without risks for caribou because we estimate that stands managed this way will, in time, have about half of the available lichen biomass of an uncut old forest (Caribou Strategy Committee 1996). Evidence suggests that caribou will still use stands with these biomass levels. Additionally, immediate postharvest lichen growth rates are up to 15% lower than in an uncut old forest (Fig. 3). Despite these concerns, we feel this approach has the best chance of success in the integrated management environment in which it is recommended.

"Modified Harvesting" Approach on Arboreal Lichen Sites >45% Slope

Stands with a slope of >45% are used less by caribou and are more difficult to harvest with a group selection system. Highlead or helicopter logging could be used to implement the logging prescription on these slopes but its value is reduced because caribou make less use of steep slopes than slopes <45% (Fig. 2). Therefore, clearcutting with reserves on slopes >45% is acceptable. To allow for some caribou use and stay within the impact modelled in the integration report (i.e., 100% of the stand is available for harvest in a 240-year period), 50% of each cutblock should be clearcut every 120 years. This strategy will result in half of each stand being 120–240 years of age at all times, thereby providing some lichen bearing habitat. This approach acknowledges that 50% of each stand will be unsuitable habitat for caribou through substantial periods of the rotation. The harvesting should be applied in the following manner:

Maximize the size of cutblocks as allowed in this NDT in the FPC *Biodiversity Guidebook*.

Harvest using clearcutting leaving 50% of the cutblock in reserves.

Keep the clearcut parts of the block as narrow as logistically possible.

Plan the cutblock so that the 50% reserve area can be logically accessed for logging in 120 years.

3. Early Winter Range outside the CCLUP Caribou Area

Some caribou use mature stands in the ICH during the early winter when these lower elevation sites have less snow than the Engelmann Spruce–Subalpine Fir (ESSF) zone. Later in the winter, as the density of snow increases, they move up into the ESSF zone. These early winter range areas should be managed to emulate the natural disturbance pattern and structure as allowed in the FPC *Biodiversity Guidebook* and landscape unit emphasis.

Radiotelemetry data show that a few specific areas are especially heavily used as early winter habitat in the ICH zone (Fig. 4). We recommend that these heavily used areas should overlap with the old seral areas of the biodiversity strategy. If harvesting must take place we recommend the group selection system. Unfortunately, the CCLUP does not provide for extended rotations for caribou in the ICH zone because early winter range in the ICH is part of mountain caribou range in the Cariboo Forest Region that the CCLUP has zoned as "conventional" harvest with regard to caribou.

Adaptive Management

The recommended harvesting approaches reflect the best current strategy given higher level plan direction, knowledge of caribou needs, ecology of arboreal forage lichens, silviculture of Engelmann spruce and subalpine fir, and logistics of timber harvesting. Many long-term questions remain to be answered and research installations will continue to be monitored. Some questions can only be answered at other scales of investigation. For example, how would caribou respond to landscape-level development using "modified" harvesting approaches? To address this and related questions, a major adaptive management trial is being planned in the Mount Tom area of the Quesnel Forest District. This trial will involve full development of about 1,700 ha while approximately 2,300 ha will remain as an uncut control. Pretreatment radiotelemetry data show substantial caribou use in the area and radiotelemetry will be expanded by monitoring more animals in the posttreatment period. Timber harvesting on this trial is scheduled to begin later this year.

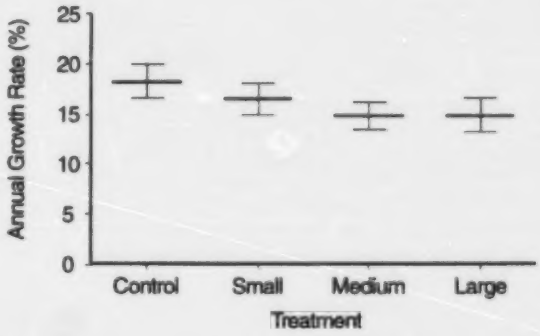

Figure 3. Annual growth rates and 95% confidence intervals of *Bryoria glabra/B. fuscescens* in 3 replicates of group selection treatments and controls in the Engelmann Spruce-Subalpine Fir zone of the Cariboo Forest Region, British Columbia, 1994. Small openings are 0.03 ha, medium openings are 0.13 ha, large openings are 1.0 ha, and controls are uncut. All treatments involved 30% volume removal.

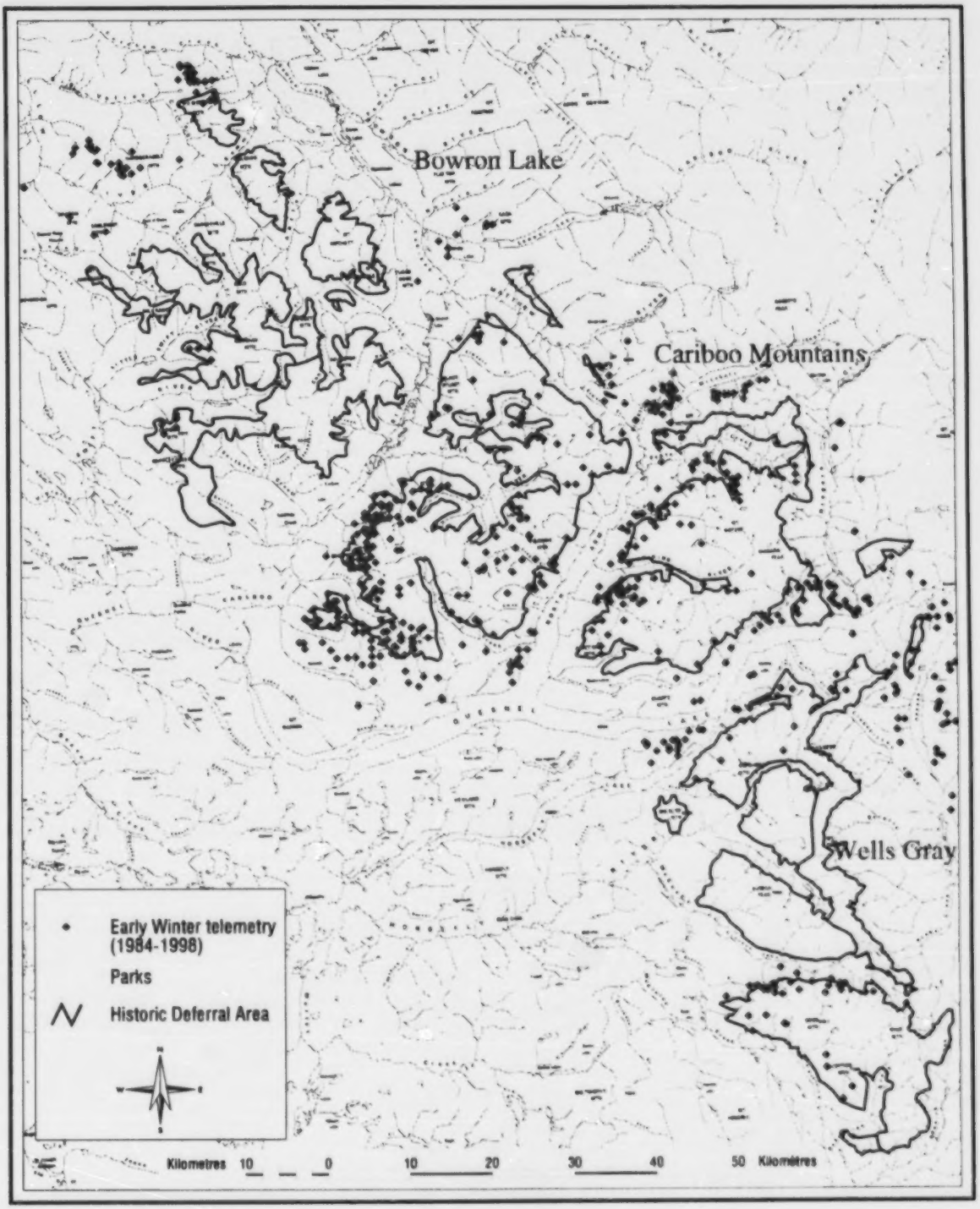

Figure 4. Early winter radiotelemetry relocations (1984-1998) showing areas of use outside of the deferral areas for which the Cariboo-Chilcotin Land-Use Plan has not made provision.

ACCESS

Two access issues are of special concern to caribou—roads and off-road motorized recreation. To address road access we recommend aggregating timber harvesting in specific landscape units over short time periods followed by deactivating roads and not accessing the area again for 20 years. Access control points are recommended for each drainage. Also, by developing caribou habitat based on even timber flow, massive harvesting will be prevented throughout the range of these animals in any single time period.

Our radiotelemetry data have shown that caribou do not use highly suitable habitat when intensive snowmobiling activity occurs in high elevation areas. A strategy is being developed to protect caribou from the negative impacts of snowmobiling by zoning caribou range into 3 categories of use: (1) unrestricted access, (2) careful access with rules to minimize the impact on caribou), and (3) no motorized access. This type of technical input is being provided to subregional planning tables to incorporate into higher level direction. This topic remains a hotly debated issue among user groups.

PREDATION

Wolf predation can dramatically reduce caribou populations where the wolf population is sustained by other prey species (Seip 1992*b*). This has occurred with mountain caribou in the Cariboo Forest Region. During the 1980s population levels decreased substantially; however, during the past 10 years the caribou population has been stable with enough recruitment to offset mortality. Legal hunting has been closed for mountain caribou in the Cariboo Forest Region to eliminate that source of mortality. The timber harvesting approaches for slopes <45% were designed not to enhance moose habitat, thus limiting prey species abundance for wolves and controlling predation on caribou. We recommend minimizing the number of snow-ploughed roads in caribou winter range to not enhance mobility of wolves. Also, the Ministry of Environment, Lands and Parks is proposing to manage moose populations in mountain caribou range to levels which will not increase wolf numbers above present levels.

Recent data collected on 2 wolf packs in the range of mountain caribou in the region indicate an elevational separation from caribou during most of the year (Fig. 5). During the caribou decline in the 1980s, the 2 species had more elevational overlap (Seip 1992*a*). Monitoring of caribou and wolf populations will continue. If current measures are inadequate for maintaining viable caribou numbers then other options will be explored.

OUTSTANDING ISSUES

To successfully maintain caribou, we need to manage many issues within mountain caribou range. These issues include forest management systems (within and outside deferrals), hunting (legal and illegal), predation, and access. The strategy has addressed all of these issues but decision-makers still need to accept the whole package. While questions remain about all these issues, major unresolved areas of concern include (1) the inability to adequately address early winter range in the ICH (outside of the deferral) while forest development is targeting these areas in the short term, and (2) the lack of an accepted snowmobile access plan. Subregional planning must keep pace with these issues and make appropriate decisions before key habitats are lost.

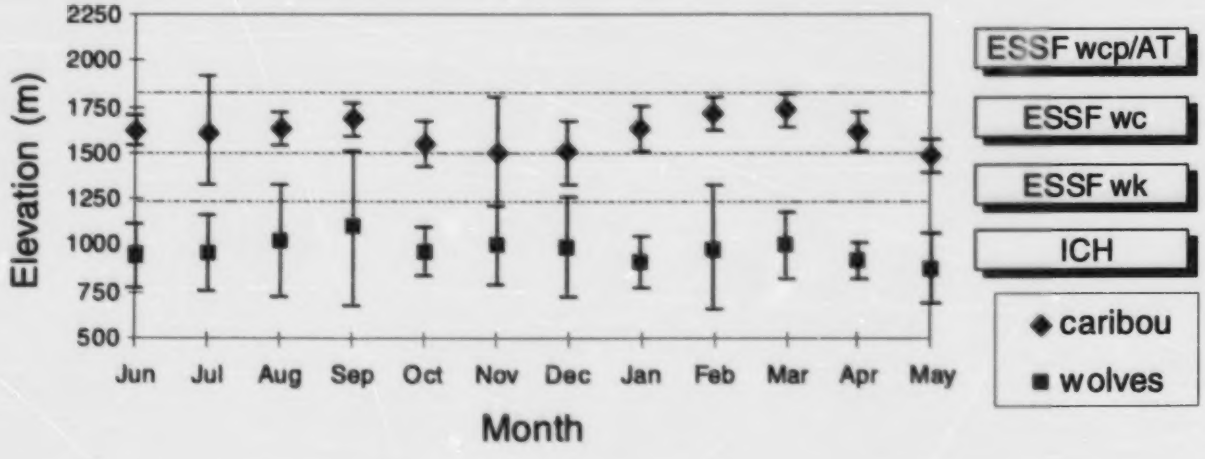

Figure 5. Mean monthly elevations used by radiotelemetry-equipped wolves and mountain caribou in the Cariboo Forest Region, British Columbia, 1993–1998. Error bars indicate ±1SD.

ACKNOWLEDGEMENTS

The strategy was developed by the Caribou Strategy Committee whose members also include M. Folkema, R. Hoffos, H. Knezevich, M. Lloyd, and M. Pelchat. Much of the strategy was developed from the results of research and monitoring by many people including M. Waterhouse, L. Roorda, D. Seip, T. Newsome, S. Walker, L. Davis, and W. Beaulieu. M. Waterhouse reviewed an earlier version of the manuscript.

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Multivariate Habitat Modeling for Mountain Caribou in the Columbia Mountains, British Columbia

C.D. Apps
Aspen Wildlife Research
2331 7th Avenue NW, Calgary, Alberta, T2N 1A1, Canada
aspen@cadvision.com

B.N. McLellan
British Columbia Ministry of Forests
RPO #3, PO Box 9158, Revelstoke, BC, V0E 3K0, Canada

J. Flaa
Parks Canada
PO Box 350, Revelstoke, BC, V0E 2S0, Canada

ABSTRACT

Mountain caribou, an ecotype of woodland caribou (*Rangifer tarandus caribou*), are associated with late successional forests in southeast British Columbia. Because they are listed as "vulnerable" in this province, measures for habitat management must be integrated at all levels of forest planning. We collected 2519 radio-locations for 49 animals over a 6-year period. Using univariate and multivariate logistic regression, we analyzed these data against overstory and terrain attributes to develop predictive spatial habitat models for each of 4 caribou seasons and for all seasons combined. At the stand-level, models were based on overstory and terrain attributes. At the landscape level, models were based on caribou selection for the dispersion of suitable stand and terrain conditions over 3 spatial scales. Seasonal models showed significant discriminating power in classifying caribou and random locations at both levels. Optimal model performance was achieved in classifying caribou habitat at $P > 0.6$ for all seasons at the stand level, and from $P > 0.4$ to $P > 0.6$ at the landscape level. Linked to a GIS, these models may serve as decision-support tools for forest management at strategic planning levels. However, underlying assumptions and limitations must be understood in their management application.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Population Status and Mortality of Mountain Caribou in the Southern Purcell Mountains, British Columbia

Trevor A. Kinley
Sylvan Consulting Ltd
Box 69, RR1, Toby Benches, Invermere, BC, V0A 1K0, Canada
sylvan@rockies.net

Clayton D. Apps
Aspen Wildlife Research
2331 7th Avenue N.W., Calgary, AB, T2N 1A1, Canada

ABSTRACT

Population surveys and mortality monitoring for mountain caribou, an ecotype of woodland caribou (*Rangifer tarandus caribou*), were conducted in the southern Purcell Mountains of British Columbia from 1994 to 1998. Results indicated that this subpopulation was declining, with annual growth rates ranging from 0.82 to 0.88. This was a result of low calf recruitment (late-winter ratios of 0.00–0.13 calves per adult) and high adult mortality (mean annual rate = 0.27). Reasons for low recruitment were unknown, but most adult mortality was attributed to predation, particularly by cougars (*Felis concolor*). The difference in mortality rate between sexes was nonsignificant ($P = 0.14$), but was consistent with increasing bull:cow ratios observed during the study. The southern portion of the study area had more recent cutblocks and fires (i.e., forests ≤40 years), greater road density, and more fragmentation than the northern portion. Caribou mortality was also significantly greater in the south than the north ($P = 0.03$). This mortality may have been partly due to post-logging changes in the distribution of ungulates favouring edges and early-seral forests, and the predators that were attracted to these ungulates. If current trends continue, the southern Purcell caribou herd will be extirpated within a decade. To address this problem, we recommend that (1) this subpopulation be augmented with animals from other mountain caribou herds; (2) relationships between predation and patterns of forest harvesting be investigated, with harvesting either deferred in some areas or designed to minimize enhancement of other ungulates' forage; and (3) limiting predation on caribou by reducing the number of cougars and alternate prey be investigated.

Key words: caribou, cougar, *Felis concolor*, forest harvesting, mortality, population, predation, Purcell Mountains, *Rangifer tarandus caribou*.

Mountain caribou, an ecotype of woodland caribou (*Rangifer tarandus caribou*), inhabit the high-snowfall region of southeastern British Columbia, extreme northern Idaho, and extreme northeastern Washington. This ecotype is largely defined by its late-winter diet, consisting almost entirely of arboreal lichens (Stevenson and Hatler 1985). In recent years, intensive research has focused on the population status and ecology of mountain caribou, due to concerns for long-term population viability. Populations have declined over most of the historic range, with many of the remaining subpopulations occurring at low densities in disjunct distributions (Paquet 1997), and threatened by the loss of the old forests on which they depend (Simpson et al. 1997). The provincial population is estimated at about 2,500, distributed in 13 subpopulations, of which about half are markedly below the assumed carrying capacity (Simpson et al. 1997; I. Hatter, BC Environment, unpubl. data). Furthermore, mortality rates for several subpopulations may significantly exceed recruitment (Seip 1992, Compton et al. 1995, Hamilton and Herbison 1998).

The southern Purcell Mountains (Fig. 1) support one of the smaller and more isolated mountain caribou subpopulations. At the southeastern-most and driest limit of current mountain caribou range, this subpopulation occurs with a greater suite of ungulates and predators than in other areas. These include elk (*Cervus elaphus*), moose (*Alces alces*), mountain goats (*Oreamnos americanus*), mule deer (*Odocoileus hemionus*), white-tailed deer (*O. virginianus*), cougars (*Felis concolor*), lynx (*Lynx canadensis*), bobcats (*L. rufus*), wolves (*Canis lupus*), coyotes (*Canis latrans*), wolverines (*Gulo gulo*), black bears (*Ursus americanus*), and grizzly bears (*U. arctos*). High rates of woodland caribou mortality have been reported for areas with extensive forestry or high numbers of predators (Bergerud 1983, Seip 1992, Cumming and Beange 1993, Compton et al. 1995, Crete and Desrosier 1995, Cumming and Lavoie 1996). Because most of the southern Purcells are under commercial forestry tenure and support several potential predators,

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

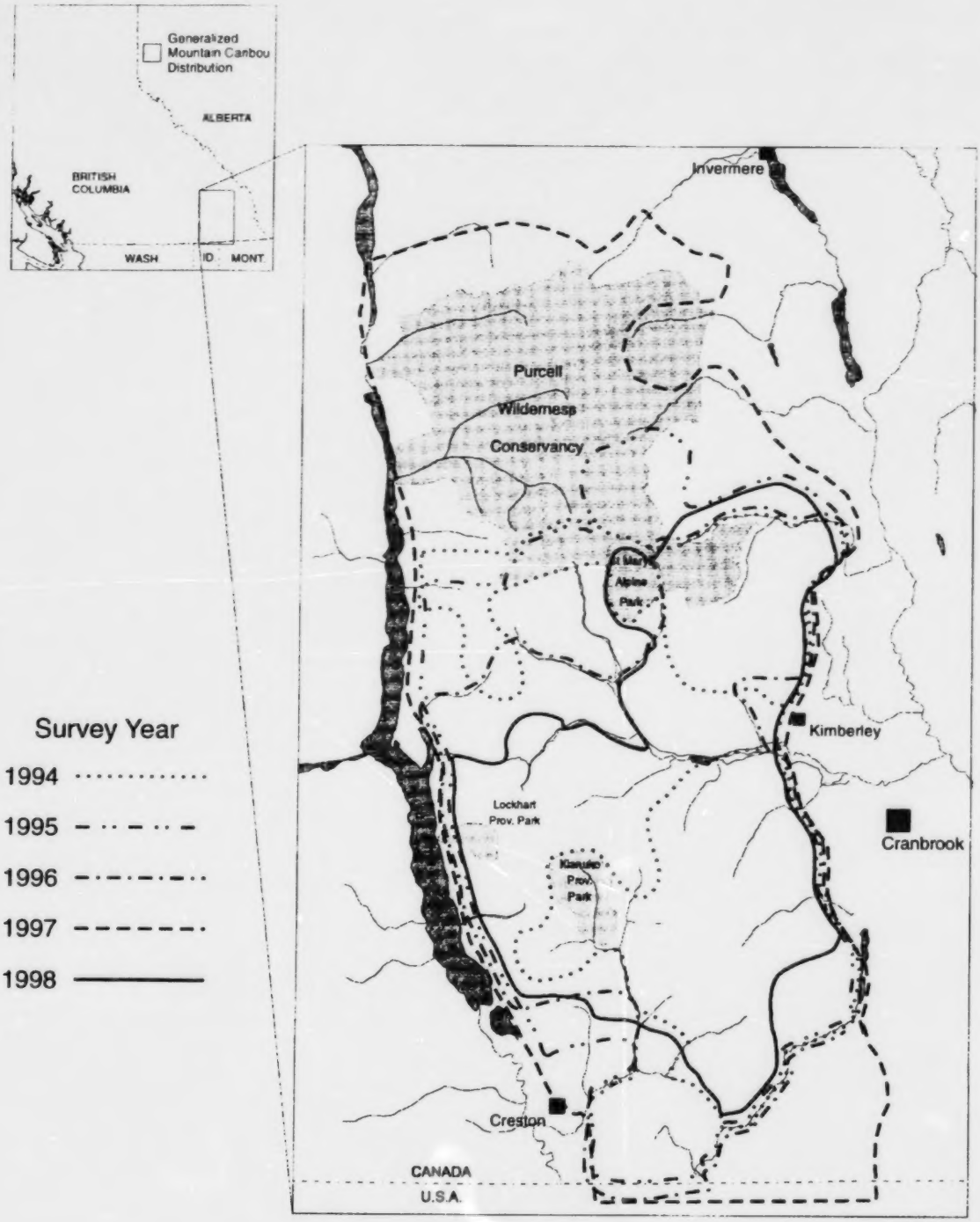

Figure. 1. Late-winter mountain caribou survey areas in the southern Purcell Mountains, British Columbia, 1994–1998.

population trends are of particular concern for this small herd. In this paper, we report results of population surveys and observed rates of mortality among radio-collared caribou. We also test for (1) gender differences in mortality rates, and (2) the potential influence of forest resource development through comparison of 2 zones with greatly differing development intensities. Management implications are discussed.

STUDY AREA

The study area covered about 6,000 km^2 near the south end of the Purcell Mountains, at elevations ranging from 530 to 2,850 m. It fell within the Columbia Mountains and Highlands ecoregion of the Southern Interior Mountains ecoprovince (Luttmerding et al. 1990). Moisture increases from east to west, and from lower to higher elevations. The Alpine Tundra biogeoclimatic zone (AT) occurs above about 1,950 m in the west and 2,100 m in the east, the Engelmann Spruce– Subalpine Fir (ESSF) zone lies at roughly 1,550–1,950 m in the west and 1,600–2,100 m in the east, the Interior Cedar–Hemlock (ICH) exists below about 1,550 m in the west, and the Montane Spruce (MS) occurs below about 1,600 m in the east (Braumandl and Curran 1992). Climax forests are typically Engelmann spruce (*Picea engelmannii*) and subalpine fir (*Abies lasiocarpa*) in the ESSF, western redcedar (*Thuja plicata*) and western hemlock (*Tsuga heterophylla*) in the ICH, and hybrid white spruce (*Picea glauca* x *engelmannii*) in the MS. Fire-successional stands of lodgepole pine (*Pinus contorta*) are common at all elevations below the AT, Douglas-fir (*Pseudotsuga menziesii*) occurs on some warmer aspects of the ICH and MS, and whitebark pine (*Pinus albicaulis*) and alpine larch (*Larix lyalli*) are common at the ESSF/AT transition. The study area contained 2 distinct zones of caribou activity, separated by the east–west trending St. Mary River Valley (Apps and Kinley 1997). Forestry development varied from nonexistent to intensive.

METHODS

Population Surveys

Adult caribou were captured in March and April of 1993, 1994, and 1997 using a net-gun fired from a helicopter, and were fitted with very high frequency (VHF) or global positioning system (GPS) collars equipped with mortality sensors. Capture efforts were geographically proportional to numbers of animals censused across the known distribution of the southern Purcell population, and samples varied from 8 to 23 animals. Schedules for aerial telemetry monitoring varied among seasons and years, ranging from twice-weekly to monthly. Caribou were surveyed in March or April of 1994 to 1998, with methods consistent with current standards (Resources Inventory Committee 1997). Within the survey area (Fig. 1), all ridges >1,800 m were examined for caribou. A pilot and 3 observers, using a Cessna 182 aircraft, flew along each side of the ridges at the elevation of the forest/subalpine parkland boundary (about 1,800–2,000 m). All caribou tracks were recorded on 1:125,000-scale topographic maps. Upon completion of these fixed-wing flights, each track location was visited with a Bell Jet Ranger II helicopter to search for caribou. The helicopter was equipped with a scanning telemetry receiver to aid in determining whether radio-collared caribou were in the groups observed, and to locate collared caribou that had not otherwise been found.

Of caribou clearly observed from the helicopter, the number of animals, age class (calf versus adult), and gender were recorded. Occasionally, fresh tracks were located but the caribou associated with them were not observed. In such cases, a minimum number of caribou were attributed to the tracks, and were included in the survey results. This estimate was applied only if tracks appeared to have been made the day of the survey, if no tracks could be found exiting the immediate area, and if tracks were separated by more than about 5 km from other areas where caribou were observed. The ratio of the total number of radio-collared caribou known to be present to the number of radio-collared caribou located without using the telemetry equipment was used as a sightability index. The number of caribou observed (collared plus uncollared, and animals plus track-only observations) was multiplied by this index to yield the estimated population size.

When "track-only" observations were included in the survey, all radio-collar frequencies were scanned to ensure that the tracks had not been made by a radio-collared caribou that had not been seen. Caribou located only with the aid of telemetry were included in the data used to calculate age and sex ratios, but were not included in the sum of animals observed. The annual rate of population change (λ) was calculated as the quotient of the estimated populations for that year and the previous year.

Mortality Rates

Upon detection of mortality during monitoring flights, carcasses were located on the ground and causal evidence was collected. This evidence included signs of sickness or starvation, predator tracks, method of kill, carcass disarticulation and caching, and presence of scat. In 2 cases, intact carcasses were removed for necropsy. A cause of death and associated confidence rating was then assigned. In this paper, data on cause of death include only mortalities for which the cause was known or considered almost certain. Annual mortality rates (q) among radio-collared caribou having known fates were calculated on a 1 May to 30 April year. A chi-square test compared differences between males and females, and between the 2 caribou activity areas ($\alpha = 0.05$). Annual survival rates, l, were calculated as $l = 1 - q$.

Habitat Fragmentation

Habitat conditions related to forestry development in the 2 caribou activity zones were quantified using a geographic information system (GIS). The 2 zones were objectively defined by assessing the adaptive kernel utilization distribution (UD) (Worton 1989) of existing telemetry and GPS collar data (T. Kinley, unpubl. data), with all locations temporally independent by ≥1 week. The minimum UD contour (97%) that encompassed all mortality locations was used to define the boundary of each zone. Forest inventory (B.C. Ministry of Forests 1995) and Terrain Resource Information Mapping (B.C. Ministry of Environment, Lands and Parks 1992) data were then used to determine the proportional occurrence of several attributes in each zone. These attributes included recent human-caused and natural disturbances (forests ≤40 years old), and road density (km/km^2). On a Boolean map of disturbed habitats (forests ≤40 years vs. those >40 years), the average centre versus neighbours (CVN) fragmentation index (Murphy 1985) was also determined. This index is based on the number of 1-ha cells different from the centre cell in a 7 x 7 moving matrix.

RESULTS

The estimated population declined from a high of 78 in 1995 to a low of 47 in 1998 (Table 1), which equates to rates of population change of 0.82–0.88 (mean λ = 0.85, standard deviation [SD] = 0.03). Calf recruitment varied annually from 0.00 to 0.13 calves/adult (Table 2), with a mean of 0.05 (SD = 0.06). Sex ratios varied from slightly more females between 1994 and 1996, to far more males in 1997 and 1998 (Table 2). Mortality occurred almost entirely in summer and fall, mostly due to cougars and other predators (Table 3). The difference in mortality rate between sexes (Table 4) was nonsignificant (χ^2_{OBS} = 2.20, df = 1, P = 0.14). Mortality was greater in the southern zone (χ^2_{OBS} = 3.67, df = 1, P = 0.03). Compared with the northern zone, the south was associated with a greater road density, a higher proportion of disturbed habitats, and a higher level of fragmentation (Table 5).

Table 1. Annual mountain caribou population and growth rates (λ), southern Purcell Mountains, British Columbia, 1994–1998.

Year	No. of observed	Collars present/ Collars observed	Population estimate	λ
1994	74	1.00	74[a]	--[b]
1995	78	1.00	78	--[b]
1996	56	1.14	64	0.82
1997	37	1.50	56	0.88
1998	27	1.75	47	0.84

[a] Incomplete census.

[b] Not possible to calculate λ because previous year's data unavailable or incomplete.

DISCUSSION

Survey areas shifted slightly each year from 1994 through 1998. These changes are unlikely to have caused a difference in results between years from 1995 to 1998 because caribou were not detected outside of the smallest survey area in any of these years. However, the 1994 survey omitted some areas in which a minor percentage of the caribou were observed in later years; therefore, the 1994 population estimate may be low.

We expected the adult survivorship rate, *l*, would be lower than the rate of change in the population, λ, because λ included calf recruitment. However, the difference between the two (0.12) was not consistent with the mean calf:adult ratio of 0.05. While a portion of the population was not directly observed, the number of calves in it was unlikely sufficient to account for this discrepancy. Similarly, immigration or misclassification were unlikely to have made up the difference. Two more realistic possibilities are (1) that sightability was higher than estimated during the latter part of the study, resulting in erroneously high population estimates in later years, or (2) the mortality rate among collared caribou was higher than among uncollared caribou.

Overall, our results suggest that the population was decreasing rapidly, due to both low calf recruitment and high adult mortality. An extension of the current λ value results in

Table 2. Late-winter mountain caribou sex and calf ratios based on uncorrected census data, southern Purcell Mountains, British Columbia, 1994–1998.

Year	Bulls	Cows	Unclassified adults	Total adults[a]	Calves	Bulls/Cow	Calves/Adult
1994	17	20	32	69	5	0.85	0.09
1995	27	39	11	77	1	0.69	0.01
1996	18	29	2	49	2	0.62	0.04
1997	21	11	8	32	4	1.91	0.13
1998	15	2	1	18	0	7.50	0.00

[a] Numbers do not match those of Table 1, because "track-only" observations are deleted in Table 2, and some animals seen and classified immediately after or before the survey are included in Table 2.

Table 3. Sources of mortality among radio-collared mountain caribou, southern Purcell Mountains, British Columbia, 1993–1997.

Cause	Jan–May	Jun	Jul	Aug	Sep	Oct	Nov	Dec	Total
Cougar predation	0	1	0	3	0	2	0	0	6
Bear predation	0	0	0	1	0	1	0	0	2
Wolverine predation	0	0	0	0	0	0	0	1	1
Vehicle collision	0	0	0	0	0	0	1	0	1
Illegal shooting	0	0	0	0	1	0	0	0	1
Fall in snow well	0	1	0	0	0	0	0	0	1
Unknown	0	1	0	2	0	0	0	0	3
Total	0	3	0	6	1	3	1	1	15

a population estimate of <10 animals in 10 years. This calculation does not account for the aging of the current population, the apparently decreasing proportion of females, the potential inability to find mates at low population densities, or inbreeding depression. Thus, a continuation of the current trend would almost certainly cause extirpation within a decade. No substantial data were available to interpret calf mortality causes, but adult mortality appeared to be largely due to predation, with cougars accounting for half of the known-cause mortalities. Almost all deaths occurred from June through October, peaking in August. High summer mortality corresponds with the results of several other populations of this ecotype in which predation was a leading cause of mortality (Seip 1992, Compton et al. 1995). This result is also consistent with data from 2 caribou that had emigrated to the Purcells from Idaho prior to the start of this study. The exact date of death was not known for these 2 animals, but was between May and July of 1991. Similarly, 2 caribou immigrating to the Purcells from Washington in summer 1996 died that season (J. Almack, Wash. Dep. Fish. and Wildl., pers. comm.).

The high mortality rates observed in this study may be attributable to several causes. Forest harvesting continually increases road density, area of regenerating cutblocks, and levels of habitat fragmentation, all of which were especially pronounced in the southern part of the study area. These effects may have changed the distribution of ungulates showing affinity to edges or early-seral stands (elk, deer, and moose), allowing them to shift into landscapes that, under a natural disturbance regime, would have been dominated by nearly contiguous tracts of old-growth forest and consequently of less value to those species. Thus, cougars and other predators may have expanded into landscapes where they would otherwise not be expected in response to a change in prey distribution. Simultaneously, caribou may have been forced into smaller habitat patches, with their movements being more predictable, thus increasing predator efficiency. A shift in cougar distribution may have been exacerbated by increased cougar populations. Although no firm cougar population estimates were available, the

Table 4. Annual mortality (q) of adult mountain caribou, southern Purcell Mountains, British Columbia, 1993–1998.

Year	Deaths 1 May to 30 April[a]					q
	Male	Female	North	South	Total	
1993/94	0/2	5/7	0/0	5/9	5/9	0.56
1994/95	1/5	2/9	0/3	3/11	3/14	0.21
1995/96	0/3	1/7	0/3	1/7	1/10	0.10
1996/97	0/2	2/6	1/3	1/5	2/8	0.25
1997/98	2/8	2/7	0/5	4/10	4/15	0.27
Mean	0.15	0.33	0.07	0.33	0.27	0.27

[a] Includes collars malfunctioning after the normal "mortality season" indicated by Table 3.

Table 5. Fragmentation measures in areas used by mountain caribou, southern Purcell Mountains, British Columbia.

Area	Forest ≥ 20 years (%)	Forest 21–40 years (%)	Total forest ≥ 40 years (%)	Road density (km/km²)	Fragmentation Index (CVN)[a]
North of St. Mary River	1	6	7	0.1	3.2
South of St. Mary River	13	10	23	1.0	8.1

[a] CVN = centre versus neighbours.

Ministry of Environment, Lands and Parks managed cougar harvest on the assumption that cougar numbers increased substantially in southeastern British Columbia through the 1990s, with regulations liberalized in the mid- and late 1990s. Furthermore, a high density of open roads may have increased the vulnerability of caribou to illegal harvest, vehicle collisions, and displacement. Although the influence of forestry development on mortality rate is hypothetical, the much higher rate of mortality observed in the more heavily developed portion of the study area is strongly suggestive; this mechanism has been proposed for another ecotype of woodland caribou by Cumming and Lavoie (1996). Simpson et al. (1994) noted that mountain caribou are abundant only in areas with 60% old-growth habitat. This finding might not always be related to habitat-mediated predation, but is at least consistent with the hypothesis.

Research may have contributed to the high mortality in the southern Purcells. For example, collared caribou may have been more obvious to predators either because of the radio-collars or the circling aircraft, or may have suffered long-term effects of capture myopathy. However, the number of dead collared animals could not account for the rate of population decrease; therefore, this explanation appears to only partially account for the observed pattern, if at all.

Trends in sex ratios are unclear. Although nonsignificant, the difference in mortality rates between males and females suggests that mortality may have been higher among females. Such a difference would be more consistent with the apparent shift in sex ratios from the first 3 years of surveys to the last 2 years. On the other hand, the degree of the apparent shift in population sex ratios was far greater than any observed mortality differences. Furthermore, small sampling errors in observations of mortality rates or in the true sex ratios among unclassified adults would have resulted in large differences in calculated proportions. A final confounding factor is that males and females might have been unequally sighted during the surveys. The 1997 survey occurred in a year of exceptionally heavy snowfall, while the 1998 census was so late that the snowpack did not support their weight. In both cases, cows and calves, having shorter legs, may have been more inclined to move downslope to the shelter of the forest where they would have been less visible to observers in aircraft. In both years, all of those collared caribou that were not observed in the census were females, which supports this possibility. Coincidentally, censuses in the nearby central Selkirk Mountains population indicated a decreasing proportion of females from 1997 to 1998 (Hamilton and Herbison 1998). This indication of a decrease in the proportion of females in the southern Purcells among such a small population warrants further investigation.

MANAGEMENT IMPLICATIONS

The data indicate that: (1) the southern Purcell caribou population declined; (2) female numbers in particular appeared to decline; (3) very high adult mortality contributed to this; (4) cougars and other predators appeared to be the major source of mortality; (5) mortality was higher in the area with greater forestry development; and (6) a continuation of the observed trend would result in extirpation within a decade.

Based on these conclusions, the following management actions are recommended:

1. Translocate mountain caribou from other subpopulations to the southern Purcells. If the Purcell herd is not augmented soon, its probability of extirpation is very high. The short-term goals of translocation would be to (a) allow the augmented population to "ride out" this potentially short-term peak in mortality rate, (b) provide more time to determine reasons for the high mortality, and (c) make the herd less susceptible to the random occurrences that influence small populations. Translocation met with little success in the southern Selkirks in past years (Compton et al. 1995), but in the most recent augmentation there, survivorship was much higher (J. Almack, Wash. Dep. Fish and Wildl., pers. comm.). Preliminary data indicate that the mortality rate in the Purcells was much lower in 1998–99 than previously (T. Kinley, unpubl. data). Furthermore, the southern Purcells offer a large area of unoccupied suitable habitat (Apps and Kinley 1997), much within provincial parks. Although the success of augmentation is unknown, we submit that the potential to reestablish a self-sustaining caribou subpopulation in the southern Purcells is worth the risk of removing relatively few animals from source subpopulations.
2. Immediately address the potential link between forestry development and mortality. Data in this paper are consistent with a pattern of high mortality in areas of heavy forest harvesting. Current research into connections between habitat fragmentation, alternate prey species, and mortality in the Selkirk Mountains should be strongly supported. Until this link is better understood, forestry development should be deferred across large areas of caribou habitat. Failing this, new forestry prescriptions should aim to minimize habitat enhancement for other ungulates.
3. Institute more detailed tracking of cougar population levels; then, if necessary, further reduce local populations of cougars and alternate prey through changes to hunting regulations. This recommendation must be approached cautiously, as it is unclear whether observed kills were caused by one or many cougars, whether a moderate increase in the harvest of adult male cougars would decrease the cougar population, or whether moderate reductions in the numbers of other ungulates sharing caribou range

would markedly influence the attraction of cougars to those areas. Such steps will only be worthwhile if a long-term commitment is made to research and to address the potential link between habitat alteration and mortality.

ACKNOWLEDGEMENTS

The Purcell Caribou Project was coordinated by the East Kootenay Environmental Society, with funding from Forest Renewal British Columbia, the Habitat Conservation Trust Fund, World Wildlife Fund Canada, and Mountain Equipment Co-op. We thank J. Bergenske, A. Levesque, and M. Panian for securing funding; L. Earl, K. Hebert, B. Burlingham, and A. Strauss for aerial surveys and telemetry; P. Rice for necropsies; and M. Jalkotzy, J. Krebs, B. McLellan, G. Mowat, and H. Schwantje for advice in interpreting causes of death.

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Effects of Partial Cutting on Arboreal Lichens Used by Mountain Caribou

Susan K. Stevenson
Silvifauna Research
101 Burden Street, Prince George, BC, V2M 2G8, Canada
sksteven@pgweb.com

Darwyn S. Coxson
University of Northern British Columbia
3333 University Way, Prince George, BC, V2N 4Z9, Canada

Key words: *Alectoria sarmentosa*, *Bryoria*, lichen, mountain caribou, partial cutting.

EXPANDED ABSTRACT

About 10 years ago, managers began to experiment with partial cutting rather than clearcutting in mountain caribou habitat. The rationale was to determine whether it was possible, through use of non-clearcutting silvicultural systems, to harvest timber and also maintain the habitat attributes that are critical to caribou (Stevenson et al. 1994). Much has been learned from those early trials about layout, harvesting practices, and short-term impacts on timber values and caribou habitat attributes (Jull et al. 1996, Cariboo Forest Region 1997, Jull and Stevenson 1999). A new generation of operational partial cutblocks is under way, building on the results of the earlier trials (Waters 1996, Armleder et al. 2000, Stevenson et al. 1999).

Recently, the move to improve biodiversity conservation by patterning forest management after the characteristic natural disturbance regime (Province of British Columbia 1995, Voller and Harrison 1998) has given added impetus to the use of partial cutting in the forests of the Interior Cedar–Hemlock and Engelmann Spruce–Subalpine Fir biogeoclimatic zones in southeastern British Columbia. Most mountain caribou range is characterized by a natural disturbance regime in which stand-destroying events are infrequent, and regeneration occurs largely through the death of individual trees or small groups of trees. Single-tree selection and group selection resemble the natural disturbance regime in these stands more than do even-aged silvicultural systems. However, no silvicultural system perfectly imitates nature. Partially cut stands typically differ from natural stands in several ways: more trees are usually removed in a harvest entry than would die naturally; few dead trees are left standing after a harvest entry; and, over time, more and more of the trees are in younger age classes.

Partial cutting can affect mountain caribou in a variety of ways. Here we discuss only the effects of partial cutting on the main winter foods of mountain caribou, the arboreal lichens *Bryoria* spp. and *Alectoria sarmentosa*.

The most immediate impact is the loss of the arboreal lichens on the trees that are felled for harvest. Sometimes caribou forage on the felled trees, but those lichens are available only briefly. The reduction in lichen biomass exceeds the level associated with merchantable timber removal, because of the additional loss of lichens on dead or dangerous trees that are felled to ensure a safe work environment.

The lichens on the remaining trees are exposed to more wind than they were exposed to before cutting. In extreme instances, due to topographic exposure, excessive timber removal, or both, much of the lichen in the residual stand may be blown off. More commonly, increased exposure may result in a pulse of litterfall after harvesting, followed by stabilization.

By opening up the canopy of a stand, partial cutting alters the canopy microclimate. We expect that at any given level in the canopy, the microclimate will be slightly windier, drier, and more extreme in temperature in a partially cut stand than in an unharvested stand. These microclimatic changes are likely to affect the physiological activity, growth rates, and fragmentation rates of the lichens in the canopy. So far, our studies of growth rates of lichens in the lower canopy of Engelmann spruce–subalpine fir stands indicate a decline in the growth rate of *A. sarmentosa*, but not necessarily of *Bryoria* spp., after partial cutting. These results suggest that a gradual shift in genus composition will occur, and are consistent with the stratification of epiphyte species in a forest canopy along a moisture gradient described by McCune (1993). As caribou select *Bryoria* spp. over *A. sarmentosa* in feeding trials (Rominger et al. 1996), such a shift in genus composition may not be unfavourable to them.

Distances between old trees and many of the young trees will be greater in a partially cut stand than in an unharvested stand. Studies of dispersal of lichen propagules (Stevenson 1988, Dettki 1998) have shown that *Bryoria* spp. disperse effectively over much greater distances than *A. sarmentosa*. *Bryoria* colonization is likely to exceed background levels throughout the openings in a partial cut, but not necessarily in a clearcut. *A. sarmentosa* may be uncommon on young trees in the interior of openings, especially if

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it is limited to the lower canopy in the adjacent mature trees.

Because of the altered age structure in a partially cut stand, a higher proportion of the substrate available for lichen growth will be young. Young branches support lower biomass of *A. sarmentosa* and *Bryoria* spp. than do old branches, even when the effect of branch size is controlled (Esseen et al. 1996). This may occur in part because *Bryoria* spp. grow more abundantly on the defoliated portions of branches than on the needle-bearing portions, and the defoliated parts constitute a higher proportion of old than young branches (Goward 1998).

It seems clear that caribou-forage lichens will still be present in interior wet-belt stands after partial cutting, that the total amount of available forage will be lower in partially cut stands than in uncut stands, and that the relative proportion of *Bryoria* spp. to *A. sarmentosa* will increase. There are many unanswered questions about the dynamics, magnitude, and functional processes associated with these changes. To address these questions, we are currently investigating the effects of the size and pattern of selection harvest openings on distribution and abundance, physiological functioning, growth and fragmentation, and litterfall rates of arboreal caribou-forage lichens in wet-belt forests east of Prince George.

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Fire, Terrestrial Lichens, and the Itcha-Ilgachuz Caribou

Trevor Goward[1]
Herbarium, Department of Botany
University of British Columbia, Vancouver, BC, Canada
tgoward@mail.wellsgrey.net150

ABSTRACT

This paper presents 11 testable hypotheses pertaining to terrestrial lichens, forest dynamics, and woodland caribou on the Chilcotin Plateau of south-central British Columbia. Based on preliminary studies conducted in the lodgepole pine forests of the Very Dry, Cold subzone of the Sub-boreal Pine–Spruce biogeoclimatic zone (SBPSxc), it is suggested that caribou and terrestrial forage lichens are linked in a positive-feedback continuum dependent in the long term on periodic surface fire. The possibility is raised that fire suppression may adversely affect woodland caribou in this subzone. Further work is required.

Key words: caribou, Chilcotin, foraging, fire, terrestrial lichens.

As a result of increasing pressure from forestry interests, the lodgepole pine forests of British Columbia's Chilcotin Plateau have been the subject of considerable debate in recent years. At issue is the sustainability of the Itcha-Ilgachuz caribou. During the winter months these animals derive a significant portion of their carbohydrate intake from lichens (Cichowski 1989). In many regions, both arboreal lichens (*Bryoria*) and terrestrial lichens (*Cladina*, *Cladonia*, *Stereocaulon*) are utilized, though in the driest subzone of the Sub-boreal Pine–Spruce zone (SBPSxc), only terrestrial lichens are available for consumption (Goward et al. 1999).

Several studies have been conducted on caribou-lichen relations in the Itcha-Ilgachuz area, though none has been published and only 2 pertain specifically to the SBPSxc (Cichowski 1989, Miège and Goward 1999). For the present purposes, however, the most important study is certainly that of Cichowski (1989), who has shown that: 1) caribou make use of this subzone during the winter months; 2) foraging caribou exhibit a strong preference for open, mature pine stands with a canopy cover of <10–15%; 3) *Cladonia*, *Stereocaulon*, and, especially, *Cladina* are more highly sought after by caribou than other terrestrial lichens; and 4) caribou are able to detect the presence of these lichens beneath the winter snowpack—presumably as a result of a highly developed sense of smell. All of these observations are pertinent to the present study.

This paper has 2 primary objectives: to advance various "propositions" concerning lichen succession and caribou use in undisturbed lodgepole pine forests in the Very Dry, Cold subzone of the Sub-boreal Pine–Spruce zone; and to formulate these propositions into a general hypothesis on the relationships among terrestrial lichens, caribou, and their habitat.

STUDY AREA

The study area is located at 52°29'N, 124°25'W on the Chilcotin Plateau, near the Itcha-Ilgachuz Mountains in the Fraser Plateau ecoregion (Demarchi 1993), roughly 40 km northwest of Puntzi Lake, and at an elevation of 1,200–1,300 m above sea level. Topographic relief is minimal, reflecting the plateau basalts that underlie the region. Bedrock, however, is largely obscured by a deep mantle of locally derived glacial tills and moraines. Soils are generally coarse-textured and weakly developed.

The climate can be characterized as cold and dry. Average mean monthly temperatures vary between -12°C in January and 12°C in July (Steen and Coupé 1997). Frost has been recorded in every month. Reflecting the existence of a strong rain shadow in the lee of the Coast Mountains, precipitation is much lower here than at similar elevations elsewhere in the province. At nearby Puntzi Mountain, for example, annual precipitation is only 317 mm. During the winter months, snowpacks accumulate to depths of about 50 cm.

Forest cover is rather uniform in the SBPSxc. In mesic sites, the tree layer consists entirely of lodgepole pine (*Pinus contorta*), with an understory comprising dwarf juniper (*Juniperus communis*), soopalailie (*Shepherdia canadensis*), and kinnikinnick (*Arctostaphylos uva-ursi*), among others. The ground layer is composed primarily of macrolichens, including Icelandmoss (*Cetraria ericetorum*),

[1] Mailing address: Enlichened Consulting, Edgewood Blue, P.O. Box 131, Clearwater, BC V0E 1N0, Canada

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 320pp.

reindeer lichens (mostly *Cladina arbusucula*), pixie-cup lichens (mostly *Cladonia ecmocyna*, *C. cervicornis*, *C. cornuta*, *C. gracilis*, and *C. phyllophora*), pelt lichens (*Peltigera* spp.), and foam lichens (mostly *Stereocaulon alpinum*).

METHODS

The observations upon which this paper are based were made during the development phase of a sampling protocol for the rapid assessment of caribou habitat in the SBPSxc (Goward et al. 1999). Field work was conducted during August 1998, and yielded 24 plots, each consisting of 10 subplots. The subplots measured 4 X 4 m, and were laid out along a transect at 8-m intervals in forest stands judged to be vegetationally homogeneous. The plots were evenly distributed among 3 forest age-classes: 40–60 years, 70–100 years, and >150 years. The raw data resulting from these plots are presented in Goward et al. (1999) and will not be repeated here.

RESULTS AND DISCUSSION

In this section, 11 testable "propositions" will be advanced concerning the ecological relationships among terrestrial lichens, caribou, and the pine woodlands that support them. These propositions have been derived from the field work reported in detail in Goward et al. (1999).

Proposition 1

At genus level, *Cladina*, *Cladonia*, and *Stereocaulon* are not limited in undisturbed pine forests of the SBPSxc by extremes of within-stand lighting.

Goward et al. (1999) found that cover values in the forage lichen genera *Cladina*, *Cladonia*, and *Stereocaulon* vary little with differing degrees of exposure to light. For example, no consistent cover value differences were observed between pine stands spaced at 0.5 m and stands spaced at 4 m. Only at spacings of >4 m were significant trends observed (i.e., a distinct increase in *Cladina* and *Stereocaulon* versus a corresponding decrease in *Cladonia*); but such trends can probably be attributed to the foraging behaviour of caribou (see Proposition 5), and are not necessarily to be construed as an ecophysiological response to exposure.

Proposition 2

Tree spacing in mesic pine stands is positively correlated with stand age.

As a rule, lodgepole pine stands become increasingly open as they mature. In the SBPSxc, Goward et al. (1999) found that tree spacing in 50-year-old forests varies between 0.5 m and 2–3 m, whereas spacing in 200-year-old forests varies between 2 m and 4–5 m.

Proposition 3

The perimeters of *Cladina* and *Stereocaulon* colonies differ markedly between open stands (in which thalli with uneven, or indeterminate, perimeters predominate) and closed stands (in which thalli with even, or determinate, perimeters predominate).

Cladina and *Stereocaulon* characteristically take the form of roughly circular colonies. Generally the perimeters of such colonies are well delimited ("determinate"), though in some cases a more ragged ("indeterminate") perimeter develops. In the SBPSxc, at least, indeterminate colonies predominate in forest openings, yet are comparatively rare in closed forest stands in which tree spacing is <3–4 m.

Proposition 4

The existence of indeterminate colony perimeters in *Cladina* and *Stereocaulon* can be attributed largely to the foraging behaviour of caribou.

Determinate colony outlines in *Cladina* and *Stereocaulon* can probably be taken as "standard" for the genera. By contrast, indeterminate outlines appear to reflect disturbance in 1 form or another. Potential disturbance mechanisms in the SBPSxc might include seasonal "churning" (Yarranton 1975), trampling, hailstones, and the ground-feeding activities of birds. None of these phenomena, however, accord with the striking observation (Goward unpubl.) that indeterminate colonies occur most frequently within the distribution area of caribou. If only for this reason, caribou can be assumed to represent a key element in the development of indeterminate outlines. Foraging caribou cause lichen fragments to be displaced outward beyond the margins of the lichen colonies being foraged (H. Armleder, B.C. Ministry of Forests, pers. comm.). Casual observation suggests that many of these fragments soon establish as small "satellite" colonies around the larger "parent" colony. Presumably it is the eventual merging of such colonies with the parent colony that confers upon the latter a ragged, indeterminate outline.

Proposition 5

Foraging by caribou favours the rapid development of *Cladina* and *Stereocaulon* "supercolonies" (to >75 cm in diamter).

The merging of parent colonies with their satellite colonies results in an accelerated rate of expansion for the population as a whole. This accelerated growth leads to the development of lichen "supercolonies"—indeterminate lichen colonies much larger than unforaged, determinate colonies growing nearby. Because supercolonies are probably highly attractive to foraging caribou, their accelerated growth rates are doubtless maintained over long periods. In the SBPSxc, *Cladina* supercolonies have been measured at up to 6 m across; by contrast, determinate colonies seldom attain a diameter of >25–30 cm.

Proposition 6
Two alternative successional pathways can be discerned in the SBPSxc: one confined to densely stocked stands, and dominated by *Cladonia*, and the other restricted to open stands, and dominated by *Cladina* and, to a lesser extent, *Stereocaulon*.

Apparently related to the foraging behaviour of caribou, stand-level terrestrial lichen succession in the SBPSxc tends to follow 1 of 2 alternative successional pathways. In the first pathway, characteristic of densely spaced stands, *Cladonia* dominates the forest floor at each successional stage, while *Cladina* and *Stereocaulon* are restricted to small, rather localized populations. By contrast, the second successional pathway is restricted to more open stands, where *Cladonia* predominates in early successional stages, but is later gradually replaced by extensive supercolonies of *Cladina* and, to a lesser extent, *Stereocaulon*.

Proposition 7
Expanding *Cladina* and *Stereocaulon* colonies invariably outcompete adjacent colonies of *Cladonia*.

As mentioned earlier, foraging caribou are assumed to act as an important mechanism of short-distance dispersal for *Cladina* and *Stereocaulon*. During foraging, many fragments are dispersed onto nearby lichen mats consisting of other genera. In time the resulting new colonies overgrow—and thereby kill—the lichens that support them. Such a dispersal strategy, which can be described as "scatter-and-smother," owes its success to 3 attributes of *Cladina* and *Stereocaulon*: 1) a brittle thallus; 2) an ability to proliferate laterally; and 3) an ability to maintain apical growth long after the death of the basal portions of the thallus.

Proposition 8
In the absence of disturbance, competition from kinnikinnick and other shrubs can be expected to increase over time.

Eight species of shrubs and 9 species of herbs have been recorded as ground cover in the SBPSxc (Goward et al. 1999). Of these, however, only kinnikinnick and, to a lesser extent, dwarf juniper vigorously compete for space with terrestrial lichens. Kinnikinnick is a much-branched, evergreen, ground-hugging shrub capable of producing solid mats over the forest floor. Expanding outwards, it successfully overrides any terrestrial lichens with which it comes into contact. No evidence of die-back has been observed in this species, whether through disease, insects, or adverse weather. Given sufficient time, then, kinnikinnick is probably an important limiting factor to terrestrial lichen cover.

Proposition 9
Stand-replacing fires generally result in densely stocked regenerating stands of little direct benefit to foraging caribou.

Lodgepole pine regeneration is strongly linked to the occurrence of wildfire (e.g., Lotan and Critchfield 1990). In the SBPSxc, stand-replacing fires generally replace old, well-spaced pine stands with densely stocked pioneer stands (Goward et al. 1999). Apparently for this reason, young pine stands are generally avoided by caribou, which have been shown to favour a more open stand structure (Cichowski 1989). Only after a prolonged period of self-thinning, usually lasting 100–120 years (Goward et al. 1999), do lodgepole pine stands become sufficiently open to attract the attention of these animals; and only then do their foraging activities begin to direct lichen succession toward a predominance of *Cladina* and *Stereocaulon*.

Proposition 10
Surface fires generally do not affect tree spacing.

In contrast to stand-replacing fires, surface fires generally do not promote the establishment of pine seedlings at high density. The reason for this is not clear, but may involve an inability on the part of seedlings to compete for moisture with the roots of pre-existing trees. The net result, in any event, is that pre-fire canopy spacing is maintained.

Proposition 11
Surface fires temporarily favour forage lichens through the creation of large openings on the forest floor suitable for lichen colonization prior to the reestablishment of kinnikinnick and other vascular shrubs at competitive levels.

Based on preliminary studies of fire scars, pine stands in the SBPSxc have in the past been subjected to surface fire at intervals of roughly 50–100 years (Goward et al. 1999). With each burn the ground vegetation is more or less returned to an early seral stage, and in the resulting temporary absence of competing vascular plants, lichen mats soon begin to dominate the forest floor. *Cladina* and *Stereocaulon*, for example, can apparently establish to impressive cover values (as "supercolonies") in as little as 40–50 years after fire (e.g., Yarranton 1975), whereas *Cladonia* achieves stand-level dominance even sooner. In the absence of further disturbance, however, this "lichen phase" of forest succession is likely to be short-lived: by about 60–100 years after fire, kinnikinnick and other ground-dwelling vascular species can begin to outcompete the lichens for space (Goward et al. 1999). In theory, this trend would then continue either until a continuous vascular cover had been achieved, or until it was interrupted by fire or other agents of disturbance.

SUMMARY HYPOTHESIS

While many of the above propositions clearly require further study, they do suggest a general hypothesis concerning the nature of terrestrial lichen ecology in the SBPSxc. In summary form, this hypothesis would posit that caribou and forage lichens (especially *Cladina* and *Stereocaulon*) are linked

in a positive-feedback continuum dependent for its maintenance on periodic surface fire or other mechanisms of disturbance. The details of this relationship are highly complex, but might be expressed as follows.

Terrestrial lichens constitute a primary winter food of caribou in open lodgepole pine stands throughout the SBPSxc. Foraging by these animals causes fragments of *Cladina* and *Stereocaulon* to become scattered beyond the perimeter of the colonies being foraged. Many of the fragments are scattered onto adjacent lichen mats consisting of other genera. Here they develop into expanding "satellite colonies" that override—and hence kill—the lichen mats on which they rest. Eventually the satellite colonies fuse with their originating "parent" colonies, thus considerably accelerating overall colony expansion. With repeated foraging, the resulting "supercolonies" can attain diameters of several metres in as little as 40–50 years. Unforaged colonies, by contrast, seldom attain a diameter of >25–30 cm; such colonies are generally associated with densely stocked stands, in which foraging by caribou is minimal.

Young lodgepole pine stands are generally rather densely stocked, and are hence avoided by foraging caribou. Yet because stand spacing tends to increase with increasing forest age, most stands can be expected to favour foraging by caribou by 100–120 years. In the meantime, competition by encroaching vascular shrubs, especially kinnikinnick, might be expected to render the forest floor unsuitable to extensive colonization by lichens. In fact, lichen supercolonies routinely do develop in old forest stands, suggesting that colonization by kinnikinnick is interrupted from time to time, possibly as a result of surface fire. In theory, surface fire would return the forest floor to early successional status at intervals of roughly 50–100 years, thus providing a temporary "window of opportunity" for the development of terrestrial forage lichens in the absence of competition from kinnikinnick. Different from stand-replacing fires, which generally initiate densely stocked regenerating stands, surface fires maintain open stand structure, and hence favour continued use by caribou.

CONCLUSION

This paper develops the hypothesis that the development of *Cladina* and *Stereocaulon* supercolonies in the SBPSxc may in part be dependent on periodic surface fire; and that, conversely, the long-term absence of surface fire (or other agents of disturbance) strongly favours increasing competition from vascular shrubs, and hence a decline in forage lichens. Here it can be added that periodic surface fire doubtless also tends to remove accumulated humus, and thus retards the establishment of well-developed soils. Well-developed soils are much less likely than poorly developed soils to support forage lichens at high abundance (Gaare 1995).

Many of the above assertions are, of course, highly preliminary; they are presented not as a guide to future management decisions, but as a stimulant to discussion and further research. Even so, the question must be raised whether the current policy of fire suppression in the SBPSxc is not likely to result in a long-term decline in available forage lichen habitat. Ahti and Hepburn (1967) have already proposed for northern Ontario that "light ground fire in pine-lichen woodlands would not interfere with the normal development of the pine, nor reduce the value of the current timber resource to any significant degree." Perhaps future research will lead to similar recommendations for some portions of the Chilcotin Plateau. Potential implications for other land-use decisions are beyond the scope of this paper, but warrant careful deliberation.

ACKNOWLEDGEMENTS

I wish to thank H. Armleder for continued support of the studies upon which this paper is based. D. Burgess provided capable assistance in the field. H. Armleder, S. Stevenson, and P. Williston provided pertinent literature. This paper has also benefited from constructive comment by H. Armleder, A. Arsenault, D. Burgess, S. Jovan, L. Kowalsky, D. Miège, and O. Steen. The project was supported by Forest Renewal British Columbia (FRBC) and the Williams Lake office of the British Columbia Ministry of Forests.

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Telkwa Caribou Herd Recovery Project (TCHRP)

L. Vanderstar
British Columbia Ministry of Environment, Lands and Parks
Box 5000, 3726 Alfred Ave., Smithers, BC, V0J 2N0, Canada
Levander@smithers.env.gov.bc.ca

R. Keim
British Columbia Ministry of Forests
Bag 6000, 3333 Tatlow Rd., Smithers, BC, V0J 2N0, Canada

ABSTRACT

The Telkwa caribou herd suffered a serious decline in the mid 1960s, largely the result of excessive hunting. The herd numbered >271 caribou in 1965, falling to >34 in 1968. The herd grew over a period of 20 years to about 70 animals (1986 count of 68 caribou), and appeared to be on the road to recovery. However, counts from 1993 to 1997 ranged from 6 to 13 caribou with minimal calf survival. The Telkwa caribou herd was obviously under a serious threat of being extirpated. After extensive public consultation, a recovery plan was prepared by the British Columbia Ministry of Environment, Lands and Parks and put into action in 1997. To date, some 30 animals have been introduced from the Sustut-Chase herd to augment the remaining resident Telkwa caribou. 1998 calf survival was 100% (sample of 5). All relocated caribou were fitted with radio telemetry collars for tracking and research purposes. Weekly fixed-wing monitoring flights located the collared caribou and recorded their location via GPS. Telemetry location overlays illustrate seasonal range occupation in relation to the Telkwa Caribou Herd Recovery Project area. Refinement of description of high use caribou habitat is ongoing. Interim timber harvest guidelines within and adjacent to the project recovery area are near completion. Human access management zones have been established as part of recovery efforts. Copies of the TCHRP plan will be made available in addition to interim timber harvest guidelines.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two
B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Rates and Causes of Grizzly Bear Mortality in the Interior Mountains of Western North America

Bruce N. McLellan
British Columbia Ministry of Forests
RPO #3, P.O. Box 9158, Revelstoke, BC, V0E 3K0, Canada
Bruce.McLellan@gems9.gov.bc.ca

Fred W. Hovey
British Columbia Ministry of Forests
RPO #3, P.O. Box 9158, Revelstoke, BC, V0E 3K0, Canada

John G. Woods
Parks Canada
Box 350, Revelstoke, BC, V0E 2S0, Canada

ABSTRACT

Understanding rates and causes of grizzly bear (*Ursus arctos*) mortality is critical to their conservation. Using data obtained from 13 study areas in the Rocky and Columbia mountains of Alberta, British Columbia, Montana, Idaho, and Washington, we estimated survival rates and causes of mortalities for 388 grizzly bears radio-collared for research purposes between 1975 and 1997. We found that people killed 77–85% of the 99 bears that were known or suspected to have died while radio-collared. In jurisdictions that allowed grizzly bear hunting, legal harvest accounted for 39–44% of the deaths. Other major causes of mortality included control killing for threatening human habitation or property, self-defence, and malicious killings. The mortality rate due to hunting was higher for males than females, and subadult males had a higher probability of being killed as problem animals than did adult males or females. Adult females had a higher mortality rate from natural causes than males. The annual survival rates of subadult males (0.74–0.81) were lower than for other sex-age classes. Survival rates of adult males varied between 0.84 and 0.89 in most areas. Comparing the survival rates of females between areas dominated by multiple-use land with those dominated by parks, we found that although few bears were killed within park boundaries, survival rates appeared to be higher in multiple-use areas (0.95–0.96 vs. 0.91). The importance of well-managed multiple-use land to grizzly bear conservation should be recognized, and land-use plans developed for these areas should ensure there is no human settlement and only low levels of recreational activity. Without radiotelemetry, management agencies would have been aware of only about half (46–51%) of the deaths of radio-collared grizzly bears.

Key words: Alberta, British Columbia, grizzly bear, hunting, Idaho, Montana, poaching, problem wildlife, protected areas, survival rate, *Ursus arctos*, Washington.

Due to late maturation and a low reproductive rate, grizzly bear populations are most sensitive to a change in female survival (Knight and Eberhardt 1985, McLellan 1989, Eberhardt et al. 1994, Hovey and McLellan 1996, Mace and Waller 1998). Unlike most other large mammals, however, grizzlies are killed for a variety of reasons in addition to legal sport hunting. Understanding the causes and rates of grizzly bear mortality is fundamental to grizzly bear conservation, but because grizzly bears are difficult to capture and keep radio-collared, individual research projects rarely collect sufficient information on mortality factors to make general inferences. In this paper, we present the analysis of mortality data collated from several telemetry-based studies of grizzly bears from areas with a variety of management goals. We also estimated the proportion and types of grizzly bear deaths that would not have been recorded by management agencies unless the bears had been radio-collared.

METHODS

We analyzed data from 13 study areas in the Rocky and Columbia mountains of British Columbia, Alberta, Montana, Idaho, and Washington, collected between 1975 and 1997. Due to low sample sizes, data from several studies in geographically similar or adjacent areas with similar management goals were pooled. Study areas were the Mountain Parks, Waterton/Blackfeet, Northfork (NF) Flathead, Southfork (SF) Flathead, and Cabinet/Selkirks.

Bears were captured for research purposes and radio-collars were attached with a canvas connector that decomposed

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

and allowed the collar to drop from the bear after a planned amount of time. Bears killed as problem wildlife, for defence of life or property, taken legally by hunters, and some illegal killing were reported to or investigated by conservation officers. Cause of death of other radio-collared bears was determined by investigating the site.

Mortalities were first classified as natural, human-caused, or unknown. Those classified as human-caused were further categorized by the apparent reason: 1) legal hunting; 2) malicious, when the animal was shot and left for no apparent reason; 3) management problem, when the bear was near buildings, camps, or livestock, and killed or removed by a wildlife official; 4) citizen's problem, when a citizen shot the bear for being near buildings, livestock, or a camp; 5) self-defence, when a person thought their safety was threatened; 6) poached, when the animal was hunted but killed illegally; 7) accident, such as a vehicle collision; and 8) unknown, when a radio-collar had been cut off. We also recorded suspected human-caused deaths and whether or not the management agency would have recorded the death if the animal had not been radio-collared.

Survival rates were estimated for each sex-age class using Kaplan-Meier estimator described by Hovey and McLellan (1996). We tested differences in survival rates among study area groups and sex-age classes using an unbalanced, 2-way analysis of variance (ANOVA; Montgomery 1991). Differences in mortality rates associated with different causes were analyzed as a 1-way ANOVA treating sex-age class as the design factor. Because each jurisdiction had different management goals, practices, and laws, we summarized causes of mortality by jurisdiction. Because collared bears frequently moved among jurisdictions we had to calculate survival rates for each study area group.

RESULTS

Of 388 radio-collared grizzly bears that were monitored for a total of 704.4 radio-tracking years, 90 (23%) were known to have died, and 9 (2.3%) were suspected to have died, while radio-collared. Survival rates differed among sex-age classes ($F_{3,351} = 3.89$, $P = 0.009$; Table 1), but not among study area groups ($F_{3,351} = 0.690$, $P = 0.559$). The annual survival rates of adult males, adult females, and subadult females were not different, but all were greater than the survival of subadult males.

Depending on how the 9 suspected deaths and 5 deaths from unknown causes were treated, people caused 77–85% of the grizzly bear deaths. In British Columbia and Alberta grizzly bear hunting was legal, but it was only a major cause of mortality of radio-collared bears in British Columbia, where it accounted for 39–44% of the deaths. Ungulate hunters killing grizzly bears in self-defence, hunters mistaking a grizzly bear for a black bear (*Ursus americanus*), and malicious killing were major causes of grizzly bear deaths in Montana. Being shot or translocated by wildlife officials or shot by a citizen for killing livestock or being near homes or camps was a major mortality factor in several jurisdictions. Poachers rarely killed collared bears, there was no evidence of a collared bear dying after being wounded by a hunter, and people killed bears for unknown reasons in most studies.

Without the aid of radiotelemetry, management agencies would have been aware of 46–51% of radio-collared grizzly bear deaths and 54–66% of human-caused deaths. A large proportion of the collared bears in British Columbia were killed legally and reported by hunters, but even there the management agency would have recorded only 53–59% of the mortalities and 67–83% of the human-caused deaths.

Mortality rates due to hunting differed among sex-age classes ($F_{3,425} = 4.17$, $P = 0.006$; Table 2), with adult and subadult males having similar rates, which were higher than adult or subadult females. Mortality rates due to a combination of management and citizen control killing also differed among sex-age classes ($F_{3,425} = 4.06$, $P = 0.007$), with subadult males having a higher rate than the other 3 classes. Mortality rates from a combination of the clearly illegal categories of poaching, malicious killing, and killing for unknown reasons (collars cut off) did not differ among sex-age classes ($F_{3,425} = 1.89$, $P = 0.131$). Mortality rates from other human causes (accidents, misidentification, and self-defence) differed among sex-age classes ($F_{3,425} = 2.80$, $P = 0.040$). Adult males had a higher rate than adult females, as 5 adult males but 0 adult females were shot in self-defence. Natural mortality rates differed among sex-age classes ($F_{3,425} = 3.83$, $P = 0.010$), with adult females having a higher rate than adult or subadult males. Twelve females died of natural causes: 3 in rock or snow avalanches; 1 in a collapsed den; 5 apparently killed by conspecifics; and 3 by unknown causes.

DISCUSSION

Although grizzly bear hunting selects males over females and was permitted in some study area groups but not others, adult males had similar mortality rates in all areas. Survival rates of adult males in our study areas were similar to the 0.84 recorded in a hunted population on Chichagof Island (Titus and Beier 1994), but higher than the 0.75–0.80 recorded in the Susitna drainage of Alaska. The Susitna population was being intentionally reduced in an attempt to increase survival of moose (*Alces alces*) calves (S. Miller, Alaska Department of Fish and Game, pers. comm.).

Survival rates of adult females were similar to rates of adult males in the Mountain Parks and SF Flathead study areas and also similar to the 0.89–0.92 recorded for adult females in the grizzly bear reduction area of the Susitna drainage, Alaska (S. Miller, pers. comm.). Adult female survival rates in the NF Flathead and Selkirk/Yaak, however, were similar to that of females on Chichagof Island (0.96; Titus and Beier 1994), and

Table 1. Estimated annual survival rates of adult and subadult (excludes radio-collared cubs and yearlings) grizzly bears of each sex by groups of study areas in the Rocky and Columbia mountains, 1975–97.

Age/sex class	Study group	No. of bears	Mortalities [suspected]	Radio-years	Survival rate	SE
Adult Female						
	Mountain Parks	41	6	65.5	0.905	0.036
	NF Flathead	31	4	89.9	0.959	0.021
	SF Flathead	14	6	50.0	0.888	0.043
	Selkirk/Yaak	18	3	57.9	0.952	0.026
	Blackfeet/Waterton	14	0[2]	22.4	0.918	0.055
Combined		118	19[2]	285.7	0.926	0.006
Adult Male						
	Mountain Parks	50	7	55.8	0.891	0.038
	NF Flathead	24	4	35.1	0.887	0.054
	SF Flathead	12	3	25.4	0.888	0.062
	Selkirk/Yaak	18	5	27.8	0.842	0.066
	Blackfeet/Waterton	7	3	7.0	0.625	0.180
Combined		111	22	151.1	0.877	0.006
Subadult Female						
	Mountain Parks	14	1	17.8	0.954	0.045
	NF Flathead	25	3	46.5	0.935	0.036
	SF Flathead	18	5	35.2	0.872	0.054
	Selkirk/Yaak	10	1	9.5	0.929	0.070
	Blackfeet/Waterton	13	3	19.5	0.859	0.077
Combined		80	13	128.5	0.923	0.008
Subadult Male						
	Mountain Parks	29	8	24.6	0.742	0.078
	NF Flathead	36	5[4]	36.4	0.782	0.063
	SF Flathead	11	4	15.4	0.784	0.095
	Selkirk/Yaak	16	4	18.7	0.807	0.090
	Blackfeet/Waterton	17	3	12.1	0.798	0.106
Combined		109	20	107.2	0.801	0.007

appeared to be higher, although not statistically so, than in the Mountain Parks and SF Flathead.

The lack of difference or perhaps even higher adult female survival rates in some multiple-use landscapes (e.g., NF Flathead, Selkirk/Yaak) compared to areas dominated by protected areas (e.g., Mountain Parks) is an important consideration in developing conservation strategies. Although few collared bears died when inside park boundaries, bears had high mortality rates on the periphery. The high mortality rate along park boundaries is likely an indirect result of nearly 1 million people within a 1–2 hour drive, and approximately 43,000 residents and 28,000 hotel beds in occupied grizzly bear habitat of the Mountain Park study areas. Similarly, within the SF Flathead study area, Mace and Waller (1998) found bears with home ranges entirely within multiple-use areas had higher survival rates than bears that also used rural settlements or designated wilderness areas. We suggest that the long-term conservation value of protected areas is not only related to the amount and quality of habitat they contain and their bear management programs, but also the number and activities of people using the protected area and adjacent lands. Multiple-use lands remote from human population centres may be critical to the long-term conservation of grizzly bears, provided that they are managed for low-density human use.

Most radio-collared bears died because people killed them. Hunting was a significant factor only in British Columbia, where it accounted for less than half the deaths. It is probable that in more remote areas a higher proportion of grizzly bear deaths would be from legal hunting because, with less human settlement, control killing would be reduced (Miller and Chihuly 1987). Results from remote study areas in Alaska suggest that 78–100% of the human-caused deaths of radio-collared bears were from hunting (Schoen and Beier 1990, Smith and Van Daele 1991, Reynolds 1993, Sellers 1994).

Development and implementation of comprehensive access, recreation, and settlement plans are essential in occupied grizzly bear habitat to maintain a low density of people, particularly those who engage in activities that put bears at risk (McLellan 1990, Mattson et al. 1996, Mace and Waller 1997). Black bear and ungulate hunters killed a relatively

Table 2. Age/sex class of known mortalities of radio-collared grizzly bears that would have been recorded by the management agency. The number of additional known mortalities and suspected mortalities that would not have been detected if they had not been radio-collared are in ( ) and [ ], respectively.

	Age/sex class				
Cause of death	Adult female	Adult male	Subadult female	Subadult male	Total
Natural	(10)	0	(2)	(2)	(14)
Unknown	(2)	0	0	(3)	(5)
Human-caused					
Hunter Kill	2	10	0	4	16
Citizen Problem	0	(1)	1	1(2)[1]	2(3)[1]
Management Problem	1	3	1	5	10
Misidentification	1	0	1(3)	0	2(3)
Self-defence	0	4(1)	0	1	5(1)
Accident	0	(1)	1	0	1(1)
Poach	1	1(1)	1	2(1)	5(2)
Malicious	1(1)	(1)	(2)	(2)	1(6)
Unknown	(1)[3]	(1)	(2)[1]	(2)[4]	(6)[8]
Total: Human-caused	6(2)[3]	18(6)	5(7)[1]	13(7)[5]	42(22)[9]
Total: All Deaths	6(14)[3]	18(6)	5(9)[1]	13(12)[5]	42(41)[9]

high proportion of the radio-collared grizzly bears. Misidentification, self-defence, and problems associated with attractants such as garbage, food, and ungulate carcasses in hunting camps were often the reason for killing bears. Enforcement of existing rules about clean camping, and emphasis on techniques for hunting in grizzly bear country during hunter training courses and in regulation synopses may reduce the number of grizzly bear mortalities associated with big game hunting seasons.

Managers should incorporate appropriate estimates of unreported kills in estimates of acceptable harvest rates. These estimates, however, remain uncertain, but appear to depend on the amount of legal hunting and the degree that bears and people share habitat. In remote areas with legal hunting managers will likely be aware of >70% of the bears killed by people. In areas without legal hunting and where people commonly live, work, and recreate in occupied grizzly bear habitat, the unreported number of bears people kill is likely equal to the number reported.

See McLellan et al. (1999) for a more detailed version of this paper.

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Estimating Population Size of Grizzly Bears Using Hair Capture, DNA Fingerprinting, and Mark-Recapture Analysis

G. Mowat
Fish and Wildlife Division, Timberland Consultants Ltd.
Box 171, Nelson, BC, V1L 5P9, Canada
gmowat@timberland.org

C. Strobeck
Department. of Biological Sciences, University of Alberta
Edmonton, AB, T6G 2E9, Canada

ABSTRACT

Field biologists can now remotely capture a small tissue sample from an animal and identify the species and individual using DNA microsatellite analysis. We used this approach to estimate grizzly bear (*Ursus arctos*) population size in a 9866 km² area in southeast British Columbia and a 5030 km² area of southwest Alberta. We captured bears by removing hair at bait sites surrounded by a single strand of barbed wire. DNA fingerprinting of the root portion of the hair was used to identify individuals. We used the closed mark-recapture models available in the program CAPTURE (White et al. 1982, Rexstad and Burnham 1991) for estimation of population size. Model selection was based on our knowledge of bear biology, results from statistical tests available in CAPTURE, and Monte Carlo simulation. We captured 109 different bears and had 25 recaptures in 5 10-day trapping sessions in British Columbia. In Alberta we captured 37 bears and had 9 recaptures in 4 14-day sessions. We captured >1 bear at 36 sites in British Columbia and concluded that some of these bears were sows with cubs, suggesting this hair capture method may work for the entire bear population. We used model M_h in CAPTURE to estimate the population size in the British Columbia study area as 262 with a 95% confidence interval of 224 – 313, and for the southwest Alberta area as 74 with CI 60 - 100. Grizzly bear density was estimated at 27 bears /1000 km² in the British Columbia study area and 15 bears/1000 km² in the Alberta area. In the British Columbia study area, bears appeared to be more abundant in the north where human density and development was lower. In contrast, bears were found throughout the Alberta study area including the more heavily populated aspen parkland zone to the east. We believe that remote hair capture combined with DNA fingerprinting are promising techniques for estimating distribution and abundance of bears and potentially many other species. This approach is of special interest to management biologists because it can be applied to large areas of size similar to that which conservation and management decisions are made.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Brown Bear Museum Specimens as a Source of Historic Genetic Information and a Tool for Conservation

C.R. Miller
Department of Fish and Wildlife Resources
University of Idaho, Moscow, ID 83844 USA
mill7264@uidaho.edu

L.P. Waits
Department of Fish and Wildlife Resources
University of Idaho, Moscow, ID 83844 USA

ABSTRACT

While there is increasing interest in incorporating levels, patterns and changes of genetic variation into conservation efforts, fundamental problems persist regarding how to place genetic data in a meaningful context. Other populations or species are commonly used as reference groups, but these may be severely impacted, occupy vastly different environments or have distinct evolutionary histories. When available, the use of museum specimens offers a unique opportunity to circumvent these dilemmas. Genetic data from historic populations can discern if, how much, and at what rate genetic variation has been lost - knowledge that can be invaluable in assessing variability risks and deciding if actively facilitating gene flow is necessary. Brown bears, *Ursus arctos*, in the Yellowstone and Northern Continental Divide Ecosystems populations have sustained varying degrees of demographic contraction and isolation in the last 2 centuries, and current levels of heterozygosity are significantly lower than in large interconnected populations to the north. Using museum specimens collected over the last 150 years, we are characterizing changes in genetic variation over time and comparing them to historic demographic events. We are also analyzing brown bear specimens from many extinct populations south of the Canadian border to determine the range of genetic diversity that has been lost and to examine the genetic processes of population bottlenecks and extinction. In addition, these data may allow us to evaluate the power of genetic methods to detect declining populations - an approach that holds promise for monitoring extant populations. Lastly, historic genetic information can provide important guidance for reintroduction efforts by comparing the genetic composition of extinct individuals and extant individuals to choose the best source population.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

British Columbia's Grizzly Bear Conservation Strategy: Progress to Date and Future Directions

M.A. Austin

Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
PO Box 9374 STN PROV GOVT, Victoria, BC, V8W 9M4, Canada
Matt.Austin@gems7.gov.bc.ca

ABSTRACT

The Grizzly Bear Conservation Strategy (GBCS) was launched by the government of British Columbia in June, 1995 and is intended to be a comprehensive approach to the conservation of this vulnerable species. Elements of the GBCS include: 1) a commitment to establish Grizzly Bear Management Areas where grizzly bear hunting will be restricted and where habitat and access issues will be addressed through land use planning, 2) the creation of an independent Grizzly Bear Scientific Advisory Committee, 3) increased research and inventory of grizzly bear populations and habitat, 4) improved hunting regulations, 5) increased enforcement, 6) increased penalties for poaching and trading in bear parts, 7) a comprehensive educational program, 8) reducing conflicts between people and grizzly bears and 9) partnerships with the private sector to raise funds for grizzly bear conservation. In addition, other projects have been undertaken such as detailed input on grizzly bear habitat needs to land use planning processes across the province, the creation of a cross jurisdictional committee to coordinate the conservation and management of grizzly bears in the Rocky Mountains, the delineation of Grizzly Bear Population Units and recovery planning for threatened populations. A review of the progress made to date under the GBCS will be presented and future directions for the strategy will be discussed.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Biology and Conservation Challenges of Badgers in the East Kootenay Region of British Columbia

Nancy J. Newhouse
Sylvan Consulting Ltd.
Box 69, R.R.1, Toby Benches, Invermere, BC, V0A 1K0, Canada
sylvan@rockies.net

Trevor A. Kinley
Sylvan Consulting Ltd.
Box 69, R.R.1, Toby Benches, Invermere, BC, V0A 1K0, Canada

ABSTRACT

Badgers (*Taxidea taxus*) are Red-listed in British Columbia. We summarize preliminary badger home range size and habitat requirement data from the East Kootenay region of British Columbia. Home ranges were 10–200 times larger than reported from studies in the United States. Using the minimum convex polygon method, female home ranges averaged 54 km^2 and males 509 km^2. Low trap success, large home ranges, predominantly adult captures, high mortality, and low natality suggest a small population, particularly in the north. Burrows used by radio-tagged badgers had more commonly been re-used than recently excavated ($P < 0.001$). Most (77%) had Columbian ground squirrel (*Spermophilus columbianus*) burrows within 50 m, which exceeded their relative availability ($P < 0.001$). The percentage of radio-locations on nonforested land (57%) was greater than the relative availability of such habitats ($P < 0.001$), although forested land with crown closure of up to 85% was used. Badgers occurred closer to paved roads than expected ($P < 0.001$). Soils for most radio-locations were medium to moderately coarse textured, and half occurred on fluvioglacial material. All available biogeoclimatic zones were used, including the Ponderosa Pine (PP), Interior Douglas-fir (IDF), Montane Spruce (MS), Engelmann Spruce–Subalpine Fir (ESSF), and Alpine Tundra (AT), but 77% of radio-locations were in the IDF. Diet analysis revealed that both males and females consumed ground squirrels, red-backed voles, beetles, sparrows, loons, and fish. Conservation of badgers is hindered by lack of information on them and their prey, failure to recognize their role as a wide-ranging carnivore, and the degree of change occurring in their primary habitat of open forest and grassland. A successful conservation plan will require education, cooperation with private landowners, protection or enhancement of key habitat elements, and, potentially, translocation of badgers into depleted areas.

Key words: badger, East Kootenay, home range, radiotelemetry, *Taxidea taxus*.

Badgers (*Taxidea taxus*) have recently been up-listed to Red status in British Columbia, meaning that they are considered threatened or endangered (Conservation Data Centre 1999). They had previously been Blue-listed, largely because the low-elevation grasslands and open forests in which they usually live are rapidly being altered or lost through forest ingrowth, subdivisions, roads, and other developments. They are limited to south-central and southeastern British Columbia, including the Cariboo, Thompson-Nicola, Okanagan, extreme southern West Kootenay/Boundary, and the southern East Kootenay, including the upper Columbia valley (Rahme et al. 1995).

Badgers are adapted to capturing fossorial prey, which constitute their primary diet in most locations (Salt 1976, Lampe 1982). However, badgers are opportunistic feeders and supplement their diet with a wide variety of mammals, birds, eggs, reptiles, amphibians, invertebrates, and plants (Messick 1987). There has been little research done to define badger habitat requirements. Generally, they have been studied in open, often intensely agricultural landscapes (Todd 1980, Warner and Ver Steeg 1995) and shrub-steppe habitats (Messick and Hornocker 1981), although they are known to occur from below sea level to elevations higher than 3,660 m (Lindzey 1982). Despite their threatened status in British Columbia, there had been no intensive radiotelemetry-based research on badgers anywhere in Canada until now. In order to provide the information that is currently lacking, the East Kootenay Badger Project was initiated in 1995–96. The objectives were to:

- determine home range size, dispersal patterns, and habitat use patterns based on radiotelemetry data;
- develop a habitat suitability algorithm based on soil types, forest cover, and proximity to roads and houses;
- facilitate the use of this information by resource managers and provide input directly into forest harvesting plans and

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construction development plans; and
- provide public education to improve the understanding of badger ecology and to decrease the likelihood of them being intentionally killed.

STUDY AREA

The radiotelemetry portion of the study included portions of the upper Columbia and upper Kootenay valleys of southeastern British Columbia (the East Kootenay). This included the grasslands and open forests of the Ponderosa Pine (PP), Interior Douglas-fir (IDF), and Montane Spruce (MS) biogeoclimatic zones (PPdh2, IDFun, IDFdm2, and MSdk variants) from 49°30'N to 50°50'N. Monitoring extended beyond the boundaries to follow badger movements. The habitat and ground squirrel survey of random sites was limited to the IDF zone between 49°45'N and 50°50'N. Most of the study area fell within the East Kootenay Trench ecosection of the Southern Rocky Mountain Trench ecoregion, which is part of the Southern Interior Mountains ecoprovince.

METHODS

Trapping and Radio-transmitter Implants

Trap sites were identified by field-checking locations of previous sightings or known ground squirrel colonies. Badgers were trapped at burrow entrances using #1½ soft-catch leghold traps baited with meat (ground squirrels, rabbits, or beef liver) and scented with Carmen's Canine Call (R. Carmen, New Milford, PA). Traps were checked at least once daily. Trapped badgers were noosed and hand-injected with a sedative. This was initially 10 mg/kg of Telazol mixed at 100 mg/ml, but later a combination of 0.3 mg/kg of Midazolam mixed at 1.0 mg/ml and 9 mg/kg of Ketamine mixed at 100 mg/ml was used. Surgical implantation of an intraperitoneal transmitter was conducted either in the veterinary clinic or in the field (following Hoff 1998). Blood, fecal, upper premolar tooth, and hair samples were taken. When badgers were alert, they were released either at the original trap sites if the holes were still intact, or at nearby burrows. Teeth of study animals, along with those from roadkilled carcasses obtained from the British Columbia Ministry of Environment, Lands and Parks were sent to Matson's Lab (Milltown, MT) for aging.

Monitoring

Badgers were usually located weekly from April to September and either twice-monthly (years 2 and 3) or monthly (year 4) from October to March. A fixed-wing aircraft (Cessna 172) was used to radio-locate animals. After the animals were located from the air, a hand-held receiver and antenna were used by a ground-based observer to locate burrows. The UTM (universal transverse mercator) coordinates, forest cover type, and soil type were identified from provincial forest inventory planning (FIP) maps and Lacelle (1990) or Witteneben (1980). In addition, a site assessment was conducted. This described burrow type and size, cover type, distance to nearest change in cover type, presence and characteristics of trees or stumps within 2 m of burrow, proximity and type of 3 nearest roads, proximity and type of 2 nearest buildings, slope angle and aspect, distance to nearest change in slope, and number of ground squirrel and badger holes within 1 m of either side of 4 50-m, perpendicular transects originating at the burrow. When it was obvious whether burrows had been freshly dug or previously dug, they were classified as "new" or "old." The same data (except burrow age) were collected at 201 random plots in the IDF. Data reported in this document were collected from the summer of 1996 to September 1998.

Home Range Calculations

Home ranges were calculated using the program Calhome (Kie et al. 1994). Both the adaptive kernel (ADK) and the minimum convex polygon (MCP) methods were used. Only the locations where badgers were known to have moved from the previous locations were included in home range calculations. Home range was not calculated for dispersing juveniles.

Diet Analysis

Four gut samples and 14 scat samples from roadkills and study animals were sent to Pacific Identifications (Victoria, BC) for analysis. Dr. Susan Crockford used skeletal remains from the samples to compare to collections from the University of Victoria and the Royal British Columbia Museum.

RESULTS

Badger Capture and Status Summary

Twelve badgers were radio-tagged, including 6 adult males, 4 adult females, and 2 juvenile females. Ages of adults at the time of capture ranged from 1 to 10 years. Adult males weighed 7.7–11.8 kg and females weighed 5.9–8.6 kg. Both juveniles died in the year of capture (1 of apparent cougar predation and 1 of apparent starvation or coyote predation). In addition, 1 adult male was roadkilled and 1 adult female appeared to have been killed by a cougar. At the time of writing, the transmitter of 1 adult male was in mortality mode, but was under several metres of snow in the alpine zone so it had not yet been determined whether this animal was actually dead.

Home Ranges and Dispersal

Males had larger home ranges than females by roughly 10 times ($t = 2.88$, $df = 6$, $P = 0.03$; Table 1). For females, there appeared to be a continuous reduction in home range size from north to south. The 3 males in the northern part of the

Table 1. Home ranges (km²) and standard deviation of radiotagged adult badgers, listed from north to south for each sex. MCP = minimum convex polygon method, ADK = adaptive kernel method.

Identification no.	Sex	*n*	months	100% MCP	95% ADK	75% ADK
3	F	126	26	84	310	56
1	F	144	27	55	195	40
5	F	27	8	52	96	37
7	F	53	14	24	34	8
Mean F		4		54 (25)	158 (121)	35 (20)
2	M	57	11	513	819	222
4	M	75	20	765	1,119	367
6	M	37	15	695	1,184	716
12[a]	M	7	1	2	5	3
9	M	30	7	63	90	25
11[a]	M	10	3	42	65	35
Mean M		4		509 (315)	803 (501)	333 (293)

[a] Not included in calculation of mean, because of small sample (<25 radio-locations over <6 months).

study area had much larger home ranges than the 1 in the south. Of the 2 juvenile females radio-tagged, 1 dispersed 21 km south-southeast from her natal area before death, and the other dispersed 5 km east from her capture site before her death in August 1998.

Habitat Use

On 3 different occasions—in July, September, and November—1 male travelled from valley bottom (approximately 800 m) to the alpine (2,200–2,400 m). Another male also travelled to the alpine on 1 occasion in July. Radiotagged badgers used the IDFdm2, IDFun, MSdk, ESSFdk, and AT biogeoclimatic zones, which are all of the biogeoclimatic variants in and around the study area. However, 77% of the locations were in the IDF.

Badgers used old burrows at least twice as many times as they dug new ones (binomial test, $P < 0.001$; $n = 321$). Many burrows appear to be used year after year, and in 2 cases 2 badgers used the same burrow at different times.

Most badger burrows (77%; $n = 297$) had Columbian ground squirrel (*Spermophilus columbianus*) holes on at least 1 of 4 50-m transects originating at the burrow (binomial, $P < 0.001$). The proportion of telemetry locations having ground squirrels did not differ between the IDF and the ESSF/AT/MS ($\chi^2 = 0.34$, $P = 0.56$) or the PP ($\chi^2 = 0.00$, $P = 0.97$). Therefore, radio-locations from all biogeoclimatic zones were compared to random plots in the IDF, in which only 5% had ground squirrel holes ($n = 201$). There were ground squirrel burrows significantly more often near badger burrows than in a random sample of the landscape ($\chi^2 > 28.2$, $P < 0.001$).

The proportion of radio-locations falling within 50 m of a paved road differed between the IDF and both the higher elevation biogeoclimatic zones (MS/ESSF/AT; $\chi^2 = 5.0$, $P = 0.025$) and the PP ($\chi^2 = 10.5$, $P = 0.001$). Therefore, only data from radio-locations in the IDF were compared to random plots in the IDF. Significantly more of the radio-locations were within 50 m of a paved road (22%; $n = 226$) than were random plots (4%; $n = 201$; $\chi^2 > 16.82$, $P < 0.001$). The mean distance from a paved road for radio-locations was significantly less (0.7 km) than for random plots (4.6 km; $t = 4.66$, $df = 595$, $P < 0.001$). This data was not biased by the ease of locating animals from the road, because animals were first located from a fixed-wing aircraft and then tracked to the burrow on the ground.

Based on forest cover mapping within the IDF zone, there were significantly more badger radio-locations in nonforested habitats (56%; $n = 440$) than these habitats were represented in random plots (28%; $n = 201$; $\chi^2 > 24.0$, $P < 0.001$). For nonforested habitat types, the most common forest cover map classification of radio-locations was open range, followed by cultivated, urban, and "not satisfactorily restocked" (recent cutblocks).

Most of the radio-locations from forested sites (85%) were in polygons having 6–45% crown closure. There was some use (15%) in forests having 46–85% crown closure, but none were in very dense forest (86–100% crown closure). Badgers used all age classes, ranging from stands with trees in the dominant canopy layer being 1–20 years old (7.5%) to 141–250 years (10.3%).

Soil maps of Lacelle (1990) and Witteneben (1980) identify the decile of each of up to 3 soil associations present in each polygon. When 2 strongly contrasting textures are within 1 m of the surface, the subsurface texture is classified separately from the surface texture. For this analysis, only the dominant surface-layer association was considered. The highest percentage of radio-locations in the IDF zone (52%; $n = 390$) were in surface soils of silt loam and gravely silt loam, which is medium-textured. Random plots in the IDF zone were also most often on medium-textured soils (69%; $n = 201$). Moderately coarse-textured soils of fine sandy loam and gravely sandy loam were used more often by badgers

Table 2. Comparison of mean home ranges (km²) in the East Kootenay to those found in other studies, based on 100% minimum convex polygon (MCP) method.

Study location	Source	Females	Males
Idaho	Messick and Hornocker (1981)	2	2
Wyoming	Minta (1990)	3	8
Colorado	Hoff (1998)	8[a]	25[a]
Illinois	Warner and Ver Steeg (1995)	13	44
East Kootenay	this study	54	509

[a] Figures based on 95% adaptive kernel method. No calculations were presented for 100% MCP.

than they were available in the IDF zone landscape (48 vs. 29%; $\chi^2 = 20.8$, $P < 0.001$).

Diet

Of the 18 gut or scat samples, 5 had no bone or hair. These 5 may have contained meat, soil, or other material. The 13 remaining samples contained Columbian ground squirrel (5), voles appearing to be red-backed vole (*Clethrionomys gapperi*; 4), beetles (Coleoptera; 3), sparrows or a similar species (Passerinidae; 2), common loons (*Gavia immer*; 2), a small salmonid (Salmonidae; 1), and a large sucker (*Catostomus* sp.; 1). All food types occurred in both male and female samples.

DISCUSSION

The data presented in this report are preliminary. They are based on 3 years of research and only 12 radio-tagged animals. Therefore, the results must be interpreted with caution. Furthermore, the data summarized here only represent habitat use, except in cases where data are compared to random plots in the IDF zone. Habitat selection cannot be fully determined yet, as availability of the habitat attributes surveyed has not been compared with use via a GIS (geographic information system) analysis. In the final year of the project, a habitat suitability algorithm will be produced.

Home Range

Home range sizes documented in this study were 10–200 times larger than any reported in the literature, as summarized in Table 2. A dispersed prey base was probably a significant contributor to these large home ranges. Only 5% of random plots (0.785 ha) within the IDF biogeoclimatic zone had any ground squirrel burrows (Newhouse 1999). Low badger density because of mortality and low reproductive output also likely contributed to large home range size. As Apps (1996) noted for bobcats (*Lynx rufus*), another species near its range limit in the East Kootenay, it is likely that populations are limited by increased mortality and decreased fecundity, so they simply spread into a greater available space. Thus, the resulting large home ranges may not necessarily reflect total resources required by individuals. The percentage difference between the mean size of female and male home ranges also appeared to be higher here than elsewhere, being roughly tenfold here versus 1 to 3 times elsewhere (Table 2). It may be that there was so little home range overlap between females here (in fact there were probably gaps of vacant habitat) that males had to travel proportionately farther to have a home range encompassing a given number of females.

Indications of Population Density

Indications from the first 3 years of trapping and radiotelemetry suggest that the badger population in the study area was very low, particularly in the northern portion (upper Columbia valley). This statement is based on the following observations:

- Of 12 badgers trapped, only 2 were juveniles, 1 was 1 year old, and the remainder were all at least 3 years old. Juveniles should be more susceptible to trapping as they are less trap-wary and can travel great distances when dispersing from natal ranges.
- The 2 radio-tagged juveniles both died before their first winter.
- Of the 10 adults radio-tagged, 2 or 3 died. This mortality exceeded observed natality.
- In 1996, neither of the females captured appeared to have had young that year. In 1997, 3 of 4 females failed to produce young, and the other had only 1 kit with her at the time of capture, despite litter sizes in other studies of up to 5 (Lindzey 1982). None of the 3 radio-tagged adult females reproduced in 1998. Messick and Hornocker (1981) found that fecundity rose with age and the proportion of productive females of all ages in a given year averaged 57%. The females trapped in this survey were all between the ages of 3 and 6, so higher fecundity would be expected. Messick and Hornocker (1981) speculate that if badgers are induced ovulators, as suggested for other mustelids, then frequent copulation over an extended period might ensure a high conception rate. The low population density in the East Kootenay may have resulted in reduced frequency of copulation and hence low productivity.
- Despite extensive trapping efforts in 1997, no additional badgers were captured in the upper Columbia.

- Home ranges were much larger than in other studies.

In contrast, there may be a somewhat higher population in the southern portion of the study area (upper Kootenay valley). This is based on the following observations:

- In 1 location in the south, a mother and 2 kits were seen. Subsequently an adult female was roadkilled there and a juvenile female and adult male were radio-tagged. DNA evidence suggests that the roadkilled female was not the mother of the juvenile (Newhouse unpubl. data). This indicates that there had been at least 2 adult females, 2 kits, and an adult male in a single location.
- Additional unmarked badgers were sighted in the south twice in 1998 and twice in 1997, and an untagged badger set off traps in the south in 1997.
- The only productive female was captured near the south end of the study area and she had a very small home range (24 km², compared with a mean of 64 km² for the other 3 adult females).

If badger populations are more dense farther south, it could be related to the availability of northern pocket gophers (*Thomomys talpoides*), which, in the Rocky Mountain Trench, only occur south of about 49°45'N. The width of the trench increases farther south, and has more major, low-elevation tributaries, making badgers less prone to having long, linear movement patterns. Having the ability to move in 2-dimensional space (in the south) rather than essentially 1-dimensional space (in the north) could potentially make them less vulnerable to disturbance and mortality. However, more radio-tagged animals would be required to adequately assess relative population densities and reasons for differences between areas.

Habitat Use

The use of the alpine zone on 3 occasions by 1 male and on 1 occasion by another male could have been a result of searching for Columbian ground squirrel or hoary marmot (*Marmota caligata*) colonies. On 1 occasion it did appear that the male had been hunting ground squirrels in an avalanche path near to the ridge where he was radio-located. However, no site assessments were done on these locations due to the expense of travelling to them. Verbeek (1965) reported observing a badger at 3,100 m hunting a young yellow-bellied marmot (*M. flaviventris*) in Wyoming.

The high degree of re-use of burrows by badgers may be part of a predation strategy, because we also noted frequent use of badger burrows by Columbian ground squirrels. Alternately, re-using burrows might reflect badgers repeatedly occupying certain locales and simply conserving energy by not digging new holes. Whatever the reason, this pattern of use suggests that significant conservation value could be gained by protecting known burrows during forest harvesting or other developments.

The high occurrence of Columbian ground squirrel burrows near badger burrows, relative to the dearth of ground squirrel burrows found in random plots, supports the notion that ground squirrels are a primary food source. This is consistent with the appearance of ground squirrels as the most common item in scat and gut samples. Burrows that were not found in association with ground squirrels could have been the result of badgers resting in nonforaging areas en route to foraging areas, or simply using a different food source.

The fact that 22% of radio-locations within the IDF zone were within 50 m of a paved road while only 4% of IDF zone random locations were within 50 m of a paved road suggests that badgers were not avoiding roads. There may actually be an attraction to roads, as disturbed soil appears to support higher ground squirrel populations (Ketcheson and Bauer 1995), and Warner and Ver Steeg (1995) reported a high use of edges, possibly for travel lanes. Badgers do eat carrion, so the locations falling near paved roads might be partially a result of an attraction to roadkills. Regardless of the reason, there is a potential for high roadkill mortality among badgers. Since 1996, 1 radio-tagged male badger has been killed on Highway 93/95 and at least 6 untagged badgers have been killed in the East Kootenay (Newhouse unpubl. data).

One of the striking features of the data relating to cover type is that badgers were found in almost every category. This points to this species' inherent flexibility and is, to some extent, expected of a carnivore that diet analysis shows to rely on a variety of food sources. However, certain habitat types were used much more consistently than others. The higher number of nonforested, compared to forested, habitats among radio-locations probably relates to a higher prey density in nonforested areas. Likewise, forest cover polygons labelled as open range or cultivated were used extensively by badgers. The apparently high use of the urban category primarily reflected the use of golf courses or rural road rights-of-way, which are often labelled as "urban" on forest cover maps.

Almost all of the radio-locations were in moderately coarse-textured and medium-textured surface soils of silty loam and sandy loam. Half of the soils were of fluvioglacial parent material. These soils appeared to be selected, as they were used significantly more than they were available in the landscape based on random plots within the IDF biogeoclimatic zone. They are generally friable and relatively well-drained, which makes them ideal both for badgers and their fossorial prey. The finer-textured soils would probably be prone to flooding and collapse because of poor drainage, while coarse-textured soils might not have sufficient cohesion to prevent collapse.

MANAGEMENT IMPLICATIONS

Badgers in British Columbia present unique conservation challenges. Assuming that the preliminary results of this study are indicative of other locations in British Columbia,

badgers concentrate their habitat use in valley-bottom habitats, which typically are being heavily impacted by human development. Even within the low-elevation stratum (the IDF in this study) badgers appear to be disproportionately found near roadsides, which can lead to high direct mortality. Their overlap with human-inhabited areas also makes them susceptible to shooting, trapping, poisoning, and loss of prey. The badger's niche as a wide-ranging carnivore, at least in the East Kootenay, is not widely recognized. Based on studies done elsewhere, it is easy to assume that badgers have only a small home range, which can lead to the mistaken assumption that multiple sightings of badgers over a wide area represent many individuals. This may lead to unrealistically high population estimates. The use of a wide range of habitats, including forested and nonforested sites and all biogeoclimatic zones, coupled with a lack of information about Columbian ground squirrel habitat preferences, population cycles, and sensitivity to habitat alterations, makes badger habitat management difficult. However, habitat preferences identified upon completion of this project should, in theory, allow the conservation or enhancement of key habitat elements. A significant proportion of badger habitat is on private land and First Nations reserves, so education and cooperative ventures with landowners will be a key ingredient to successful conservation plans. Translocation may also be required in the northern portions of the badger's range, as some local populations may no longer be self-sustaining.

ACKNOWLEDGEMENTS

We thank L. Ingham, A. Dibb, S. Crowley, A. Levesque, and M. Panion for administrative and technical support. T. McAllister, R. Franken, H. Page, M. Kaneen, R. DeGraff, and A. Candy assisted with field work. Financial and administrative support were provided by the Columbia Basin Fish and Wildlife Compensation Program, East Kootenay Environmental Society, Forest Renewal British Columbia, British Columbia Ministry of Environment, Lands and Parks, Parks Canada, and Columbia Basin Trust.

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Fishers in British Columbia: Options for Conservation of a Blue-listed Species

R.D. Weir
Artemis Wildlife Consultant
C15 Pringle Road, Westwold, BC, V0E 3B0, Canada
artemis@bc.sympatico.ca

F. B. Courbold
Peace/Williston Fish and Wildlife Compensation Program
BC Hydro and
British Columbia Ministry of Environment, Lands and Parks
1011 - Fourth Avenue, Prince George, BC, V2L 3H9, Canada

ABSTRACT

The need for a conservation strategy for fishers in British Columbia has become increasingly apparent in recent years. Continued changes in the composition and distribution of adequate habitat, increased trapping pressure in previously inaccessible areas, combined with harvests during trapping seasons has likely increased the vulnerability of fisher populations in the province. To confound conservation problems facing this species, the effectiveness of past management strategies is unknown and relatively little is known about fisher ecology in British Columbia. These factors have contributed to the persistent classification of fisher populations in the province as "Blue-listed". Past management strategies for fishers have been very limited in scope and success. Harvest seasons for fishers were closed for 2 years during 1990-92, but the harvest of fishers (through incidental kills) remained basically unchanged. Fishers are also currently an "Identified Wildlife" species under the British Columbia Forest Practices Code. However, no guidelines have been implemented for this species. Several habitat and population conservation strategies for fishers could be implemented in British Columbia. For instance, habitats altered by forestry activities may be used successfully by fishers provided that stand level silvicultural treatments are modified to ensure the persistence of structure (e.g., overhead cover, coarse woody debris, den structures). For habitat conservation strategies to be successful, fisher habitat requirements also need to be included in higher level plans. In addition to habitat strategies, reducing or eliminating incidental kills of fishers would greatly diminish the conservation threat facing fishers. Changing kill traps that are intended for martens so that they exclude fishers (a design is currently available) would diminish the level of incidental harvest considerably and contribute substantially to the viability of fisher populations in the province.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

East Kootenay Fisher Reintroduction Program

Anna J. Fontana
British Columbia Ministry of Environment, Lands and Parks
205 Industrial Road G., Cranbrook, BC, V1C 6H3, Canada

Irene E. Teske
Box 442, Marysville, BC, V0B 1Z0, Canada
iteske@cyberlink.bc.ca

Key words: East Kootenays, fisher, *Martes pennanti*, reintroduction.

EXPANDED ABSTRACT

In 1994, the East Kootenay Fisher Reintroduction Program was initiated by the British Columbia Habitat Conservation Trust Fund to re-establish fisher (*Martes pennanti*) populations in southeastern British Columbia. The transplant area is south of Cranbrook and west of Kimberley. The overall objectives of the program are:

1. To reintroduce the fisher into vacant but suitable habitat.
2. To expand the existing range of the fisher in British Columbia.
3. To establish a connective corridor between isolated fisher populations in Montana and populations in British Columbia.
4. To provide legal trapping opportunities when the populations are dispersed and sustainable.

According to the habitat feasibility assessment conducted by Apps (1995), suitable fisher habitat exists in areas adjacent to the Rocky Mountain Trench in southeastern British Columbia. Historical trapping records for the Kootenay Region from 1931 onward were reviewed. The last fisher trapped was in 1982. No other documented evidence of this species was encountered until 1997, when a fisher was live-trapped in the Revelstoke area during a research project on wolverine.

Fishers were live-trapped near Williams Lake and transported to a holding facility in the East Kootenay. Average time in captivity, excluding pregnant females, was 43.6 days. Fishers were immobilized for data collection and radio-collaring. Upon release, the animals were monitored bimonthly until they disappeared out of the study area, lost their collar, or died.

Between 1996 and 1998, 61 adult fishers and 23 kits were released. In spring in 1997 and 1998, 23 kits were born while in captivity. The family units were released when the kits were approximately 3 months old. The annual mortality rate was approximately 20% for adult fishers and at least 50% for kits.

In 1998, 5 fishers reduced their movements and established home ranges after 2.1 months. In 1997, home ranges were established by 2 females, 2.3 months after release. In 1996, home ranges were established by 2 females, 2.7 months after release.

In 1998, once transient movements were discontinued, home ranges were determined using the 90% Minimum Convex Polygon method (Kie et al. 1996). The growing season (May–October) home range of 1 adult male was 12,470 ha (n = 15), while growing season home ranges of 2 adult females (without kits) were 3,399 ha (n = 19), and 1,403 ha (n = 18), respectively. These data correspond well with home-range data documented in Weir (1995). In 1998, movements from their release sites to the centre of their home ranges ranged from 8 to 47 km for the 3 fishers that established home ranges.

Habitat use data were collected between May and October for 3 years. The fishers primarily used conifer forests with >40% canopy closure ($\bar{\chi}$ = 53%), and open forested and riparian habitats an average of 17% each. Utilization of open habitats, such as burns, clearcuts, and alpine, averaged a total of 8%.

Of all the different forested habitat types used, 47% utilization occurred in mature forests (>121 yr old), with use of in-growth (61–120 yr old), and immature forests (<60 yr old) averaging 29% and 22%, respectively. The fishers tended to use elevations between 760 and 1,980 m, with the average being 1,340 m.

Future plans include: monitoring the success of the transplant; pursuing habitat protection in key areas; refining the fisher habitat suitability assessment; and improving public awareness of the fisher and other mustelids.

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Wolverine Ecology and Habitat Use in the North Columbia Mountains: Progress Report

John A. Krebs
Columbia Basin Fish and Wildlife Compensation Program
103 - 333 Victoria Street, Nelson, BC, V1L 4K3, Canada
john.krebs@bchydro.bc.ca

David Lewis
Columbia Basin Fish and Wildlife Compensation Program
Box 14, D3, Athalmer, BC, V0A 1A0, Canada

ABSTRACT

We are currently completing a multiyear project investigating demography and habitat use of a harvested population of wolverine (Gulo gulo) in a 7,000-km² area within the Kootenay region of British Columbia. The study area contains 10 registered traplines, 2 national parks, a major transportation corridor, 2 hydroelectric reservoirs, and active logging areas, and supports a substantial helicopter skiing industry. Wolverine are considered vulnerable by the British Columbia Wildlife Branch; consequently, vital rate data are important for evaluating management/conservation options. A total of 39 (14F, 25M) adult and subadult wolverine have been captured over 3,700 trap-nights using log-box traps baited with available road-killed wildlife. Population estimates for the core 4,000-km² study area based on 4 years of live-trapping data are 25.6 (95% CI: 15.6–55.3) and 24.0 (95% CI: 14.7–44.3) for 1996 and 1997 respectively. Six of 11 mortalities detected during the study to date have been human caused. Annual survival rate was estimated to be 0.77 (95% CI: 0.66–0.88) for all age and sex categories combined. Reproductive data are being gathered through radiotelemetry and follow-up ground investigations of breeding females after den abandonment. Three 2-kit litters have been produced in 14 adult female reproductive seasons. Four of these juveniles have been captured and implanted with radio transmitters. Natal den sites have been in the Engelmann Spruce–Subalpine Fir (ESSF) biogeoclimatic zone, and associated with woody debris and/or large boulder talus in undeveloped drainages. Home ranges of males ($\bar{\chi}$ = 1,005 km²) were significantly larger than those of females ($\bar{\chi}$ = 310 km²). At the study area scale, the distribution of wolverine use is highly clumped in 4 distinct utilization peaks. The 2 largest utilization peaks occur within Glacier and Mount Revelstoke national parks, disproportionate to their land base within the study area. The focus for the remainder of the project is on the monitoring of existing females and kits, and identification and characterization of natal dens in spring 1999. A population census using motion-sensitive cameras will be conducted March–April 1999. Final report and management recommendations will be completed in 2000–01.

Key words: Columbia Mountains, *Gulo gulo*, habitat use, population estimation, radiotelemetry, vital rates, wolverine.

Wolverine (*Gulo gulo*) have disappeared from almost half of their former range (Paquet and Hackman 1995). In Canada, COSEWIC (the Committee on the Status of Endangered Wildlife in Canada) has listed the eastern wolverine as Endangered and the western wolverine as Vulnerable, yet wolverine are the least known of Canada's large carnivores and conservation research needs are extensive (Hummel 1990). An understanding of vital rates is fundamental to the evaluation of conservation options.

In southeastern British Columbia and elsewhere in western Canada, persistence of wolverine and other wide-ranging carnivores depends on suitable habitat inside and outside protected areas. Pressures of human use in both protected and unprotected landscapes are important carnivore conservation issues, since potential mortality/fragmentation sources such as highways and railways, as well as disturbance/displacement agents such as human recreation, occur with increasing intensity within protected areas; and additional industrial (logging, hydroelectric generation, mining) and commercial (trapping, helicopter skiing) land uses occur in surrounding lands (Hummel 1990, Paquet and Hackman 1995).

Since wolverine are so wide ranging (home ranges 48–2,000 km²: Hornocker and Hash 1981, Magoun 1985, Whitman et al. 1986, Banci 1987, Hatler 1989), they present a problem of scale for managers and trappers alike. Wolverine do not exist in manageable numbers within individual traplines and, therefore, may be vulnerable to overharvest. In eastern Canada, increased human activity and

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

development has resulted in declining wolverine populations and, subsequently, listing of the species as Endangered by COSEWIC (van Zyll de Jong 1975, Dauphine 1989). Western populations of wolverine may be heading in a similar direction if measures are not taken now to protect wolverine and wolverine habitat. The juxtaposition of national parks and transportation corridors, as well as resource extraction activities and commercial trapping, within our studyarea provides an opportunity to examine how these landuses influence wolverine vital rates. Specific objectives are

1. Estimate vital (birth death) rates for wolverine within the North Columbia Mountains (NCM) of southeastern British Columbia.
2. Identify landscape/habitat characteristics and human use activities that correlate with use by wolverine.
3. Develop a population model to estimate current and potential rates of increase of wolverine.
4. Produce biologically-based management recommendations for the conservation of wolverine in the NCM.

This report presents the initial results from the first 45 months of field activity up to 31 January 1999. Data analysis is not yet complete, but preliminary results are presented.

STUDY AREA

The project area (7,000 km²) is located north of the Trans-Canada Highway between Revelstoke and Rogers Pass in the south and the Mica Dam in the north, and within Fish and Wildlife management units 4-37, 4-38, and 4-33 (Fig. 1). The core live-trapping area (4,000 km²) consisted of the western portion of the area, along the eastern side of the Revelstoke reservoir. The area is within the Northern Columbia Mountains ecoregion; biogeoclimatic zones include Interior Cedar–Hemlock (ICH; vk1, wk1), Engelmann Spruce–Subalpine Fir (ESSFvc) and Alpine Tundra (AT).

The study area encompasses 10 registered traplines, portions of 2 national parks (Mount Revelstoke [MRNP], and Glacier [GNP]), 2 tree farm licenses, and several Forest License and Small Business Forest Enterprise Program cutting areas. Trappers harvest approximately 3 wolverine per year. The main valley bottom was inundated in 1985 with the construction of the Revelstoke Dam. The Kinbasket reservoir to the east and north was formed in 1973 with the construction of the Mica Dam. Other land uses include mining in Goldstream/French Creek, and heli-skiing, snowmobiling, and ski-touring at numerous locations throughout the area.

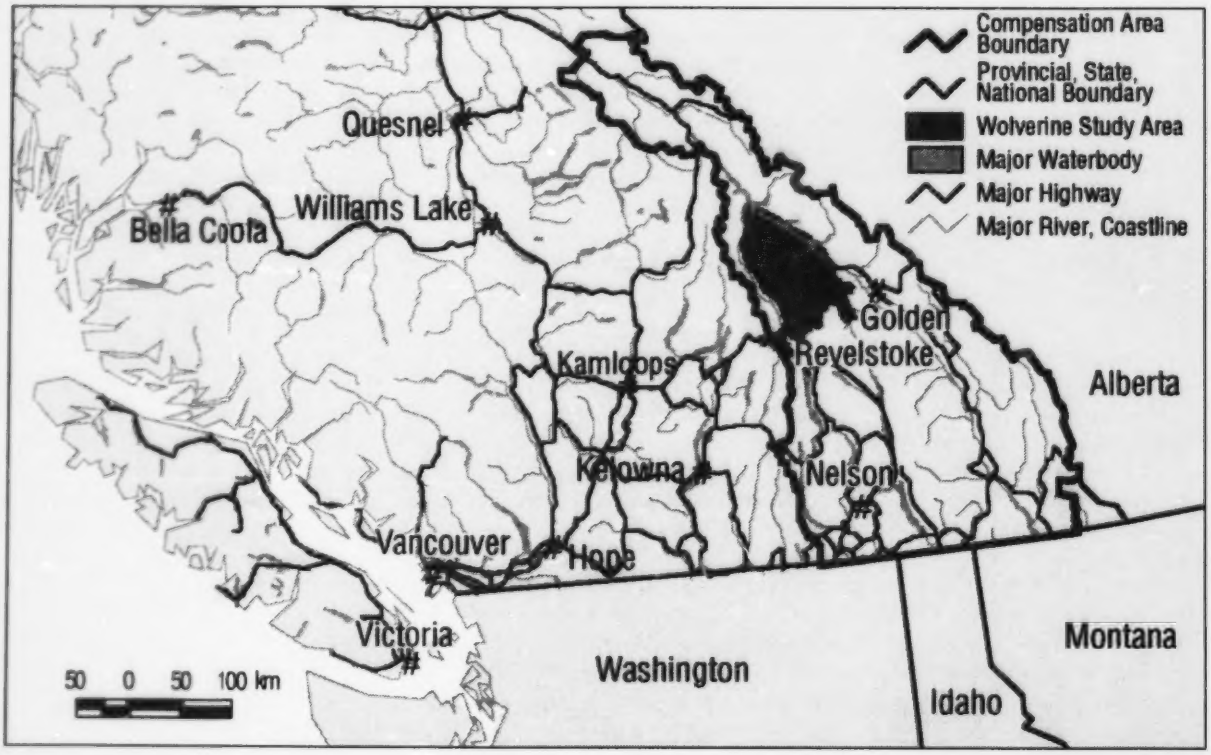

Figure 1. Location of the study area near Revelstoke, B.C.

METHODS

Live Trapping and Animal Handling

In consultation with local trappers, 33 log-cabin style and 7 portable (4 X 4 ft lumber) traps were constructed throughout the study area during 1994–97 to live-capture wolverine. Trap design was modified from Copeland et al. (1995) by using quick-release snaps as the trigger device and a bevelled front log on the door to improve fit when closed. Twenty-four traps were located in valley bottom ICHvk1 sites, 13 were within the ICHwk1, and 3 were placed in the ESSFvc. Four of the 40 traps were constructed within Mount Revelstoke and Glacier national parks. During the first winter we also used metal barrel traps (Banci 1987) opportunistically. Trapping effort was not even across the entire study area (Fig. 2), due to limitations of topography, access, and funding. Despite these limitations, our trapping effort resulted in 4 well-distributed "peaks" from north to south within the study area and was unbiased with respect to protected areas (Fig. 2).

Wolverine were immobilized with Telazol at 10mg/kg with a jabstick, ear-tagged with numbered rototags (NASCO, Modesto, CA), weighed, sexed, examined for reproductive status, and radio-collared (Lotek Engineering, Inc., Newmarket, ON; Telonics, Mesa, AZ, MOD-335). In order to minimize the chance of affecting guard hairs and underfur, a canvas insert designed to rot through after approximately 2 years was used to close the collar. The upper-left first premolar was extracted from most animals for aging purposes (Rausch and Pearson 1972). Cementum analysis was performed by Mattson's Lab in Montana.

Wolverine kits were captured at approximately 12–13 weeks of age by tracking reproductive females after emergence from natal dens. Kits were pursued on foot and immobilized after being spotted from a helicopter. Kits were surgically implanted with Telonics IMP300/L radio-transmitters under isoflourine anesthetic by an on-site veterinarian using a ventral midline procedure.

Mortality and Reproduction

Mortalities were detected via 4-hour-delay mortality sensors built into the radio-collar. We placed a high priority on recovering collars on mortality mode. Only cases where the radio-collar and the carcass were recovered were considered mortalities. Where only the collar was recovered it was

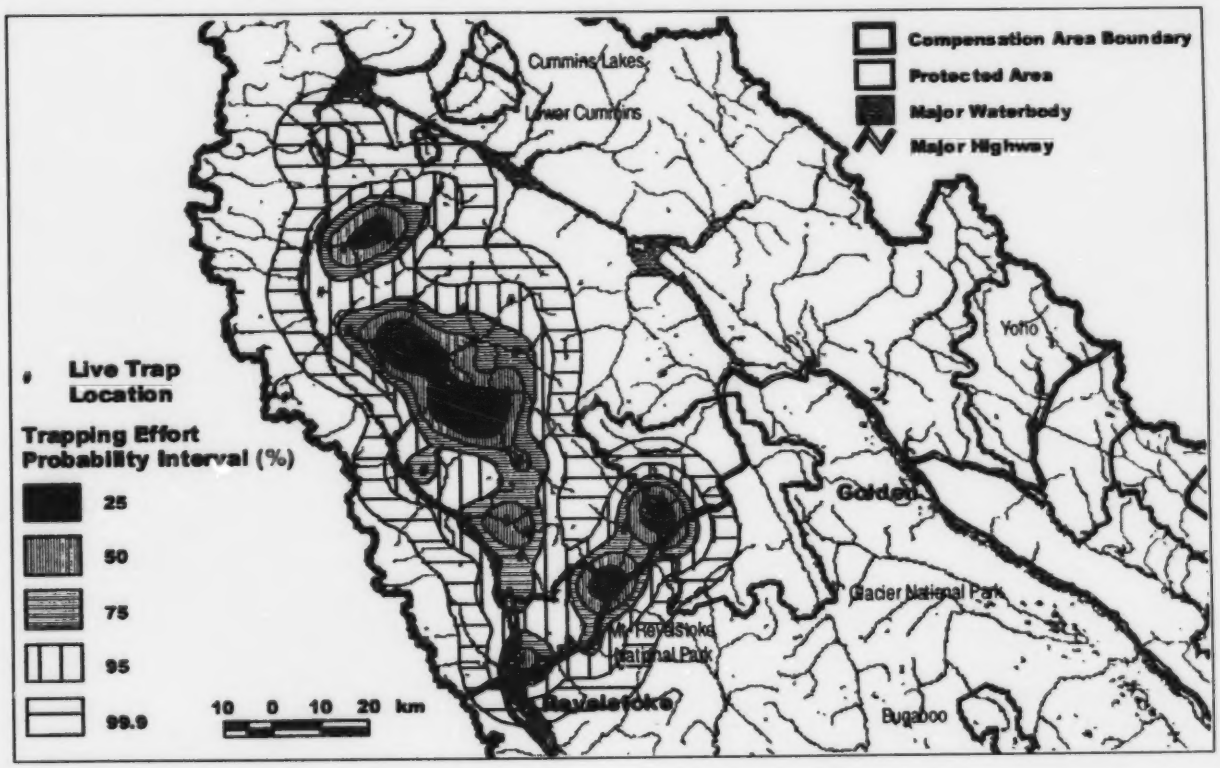

Figure 2. Adaptive kernal utilization distribution depicting live-trapping effort across the study area, 1995–98. Points correspond to trap locations.

coded as unknown. Where carcasses were recovered, a necropsy was performed by a veterinarian to determine cause of death. Survival rates were calculated following the approach outlined by Trent and Rongstad (1974) and Heisey and Fuller (1985), using program BOOTER (Hovey 1995), which bootstraps estimates to derive confidence intervals.

To confirm reproduction, radiotelemetry flights were intensified in March–late April to locate denning females. Females with identified dens were flown frequently (2 times per week or more) during May and June to increase the likelihood of obtaining visual observations of females with kits. We did not disturb den sites on the ground during the first 8–9 weeks of use to avoid potential displacement or abandonment of the site by the female.

Distribution and Habitat Use

Wolverine were located from the air using a Cessna 337 on a biweekly schedule. We attempted to locate females weekly during expected parturition dates (15 Feb–30 Apr) to establish natal den site locations. Location data collection followed that of McLellan and Flaa (1993) and included: UTM coordinates (NAD 27); forest cover polygon label (species, age, height, crown closure); habitat type (7 categories); biogeoclimatic subzone; elevation; aspect; slope; and activity of animal (if discernible). Location data precision was assumed to be ±100 m.

Location data have been presented at 2 scales: 1) as individual animal MCP (Minimum Convex Polygon) home ranges; and 2) as a representation of wolverine activity (i.e., all animals pooled) within the study area. This is useful in highlighting key locales, presumably of greatest importance to wolverine. Final habitat use analyses will treat individual wolverine separately for comparisons following multivariate procedures outlined by Aebisher et al. (1993) and Manly et al. (1993).

POPULATION ESTIMATION

Live-trapping data were used to derive an open population estimate (Seber 1982) for the 4,000-km^2 core of the study area. Each year (1995–98) was treated as a capture session in the analysis, confidence intervals were calculated following Manly (1984). An independent estimate using baited motion-sensitive cameras is planned for spring 1999.

RESULTS

Captures

Thirty-nine wolverine (14F, 25M) were live-trapped a total of 94 times in 3,700 trap-nights over 4 winters. Trapping success varied by trap type and month. Success was highest using wooden traps in February and March. Males were approximately 30% heavier than females (males 12.5 kg, n = 25, SD = 1.6; females 9.4 kg, n = 14, SD = 0.7). Ages are not yet available. Four 12-week-old male kits were also captured and implanted with radio-transmitters. Kits weighed 5.5 kg (n = 4, SD = 0.2).

Table 1. Reproductive history of radio-collared female wolverine, 1995–98. Ages determined by premolar tooth cementum analysis or estimated: subadult ≤2 yr; adult >2 yr. (D = den established but no kits survived post-weaning; DU = den established, unknown if kits survived post weaning; U = unknown; N = no reproduction; S = subadult (nonreproductive); Y (#,sex) = kits survived post-weaning.)

Wolverine	Age at capture	1995	1996	1997	1998
F204	9 (adult)	DU			
F206	1 (subadult)	S	S	N	D
F213	adult		N	N	N
F219	adult		U		
F222	adult		DU	DU	N
F223	1 (subadult)		S		
F227	subadult			S	
F228	adult			Y(2)	Y(2M)
F229	adult			Y(2M)	N
F235	subadult				S
F236	adult				N
F239	subadult				S
F241	1 (subadult)				S
F242	subadult				S
	Total adults	1	2	5	6
	Total kits	0	0	4	2
	Kits per adult	0	0	0.8	0.33

Mortality

A total of 11 radio-collared animals have died during the study: 4 were commercially trapped (M207, M217, M246, F206); 1 was killed on the Trans-Canada Highway in GNP (M212); 1 was killed on the Canadian Pacific Rail line in GNP (F223); 2 died of natural causes (F204, F229); and 2 were killed by other predators (M230, M208). One male (M224) died as a result of injuries received from either a fall or being kicked by a moose. At least 3 additional wolverine that had lost their radio-collars have also been commercially trapped (M201, F219, M232), and 1 other ear-tagged animal was recovered by backcountry skiers in MRNP (M209). Using only the radiotelemetry-located mortalities, annual survivorship for all collared wolverine up until January 1999 was 0.77 (95% CI: 0.66–0.88). The estimate for females only was similar (0.74, 95% CI: 0.55–0.93). Kit survival was slightly lower (0.61, 95% CI: 0.21–1.0), but is based on a limited sample size.

Reproduction

Three litters of 2 kits each have been produced over 14 adult female reproductive seasons (Table 1). Limited data suggest

Table 2. Characteristics of natal dens used by radio-collared wolverine, 1995–98.

Wolverine	Year	Status[a]	Dates occupied	Elevation (m)	Habitat	Structure
F204	1995	U	23 Feb–1 May	5,400	slide path	wood debris
F206[b]	1998	U	5 Apr–4 May	5,000		
F222	1996	U	13 Mar–13 Apr	5,400	mid-slope bench/ avalanche slope	wood debris
F222	1997	U	12 Mar–12 May	6,000	mature forest	wood debris
F228	1997	C	12 Mar–15 May	5,100	slide path	wood debris
F228	1998	C	16 Mar–4 May	4,900	slide path/ boulder	boulder/wood debris
F229[b]	1997	C	1 Apr–12 May	4,500		

[a] U = unconfirmed, kits not observed; C = confirmed, kits observed.
[b] Not yet ground-truthed.

reproductive rates varied greatly between years (Table 1). Females used dens during 5 seasons, but either did not produce young that survived to weaning age, or young were not detected. During the remaining 6 seasons females did not establish dens. Estimated parturition dates for the 3 litters were 1 March, 15 March, and 30 March. Characteristics and use of dens is described in a subsequent section.

Characteristics of Natal Dens

Four suspected and 3 confirmed natal den sites were found between 1995 and 1998 (Table 2). All dens were found within roadless, tributary valleys in the ESSFvc biogeoclimatic subzone under woody debris or a combination of woody debris and large boulders. Females occupied dens as early as late February and used them until mid-May in some cases. Den sites were not re-used in subsequent years. Four of the 7 den sites were located in national parks.

Home Range Size and Habitat Use

Home ranges (Table 3) of males (1,005 km^2) were significantly greater than those of females (311 km^2; $t_{1,18} = 2.85$, $P < 0.01$). Subadult males had larger average home ranges than adult males (1,611 km^2 vs. 601 km^2; $t_{1,8} = 2.18$, $P = 0.058$), whereas subadult female home ranges did not differ significantly from those of adult females (274 km^2 vs. 335 km^2; $t_{1,8} = 0.66$, $P = 0.27$). Male home ranges appear to overlap those of 1 or more females and those of other males, while ranges of females are exclusive except for accompanying young of the year and, in some cases, nonbreeding subadults. Home range boundaries were defined by geographic features (e.g., watercourses) and manmade features (e.g., reservoir, highway). Three animals crossed the Trans-Canada Highway at least 4 times; 1 was struck and killed.

Use of available habitat differed between males and females ($G = 68.7$, df = 2, $P < 0.005$) and by season. In winter, males used ICH habitats proportionately more than expected (Fig. 3; $G = 126.6$, df = 2, $P < 0.001$), whereas females used ESSF habitats more (Fig. 3; $G = 61.0$, df = 2, $P < 0.001$). In summer, males spent more time in ESSF habitats, while females used higher-elevation AT habitats (Fig. 3). The effects of human activities and land use have not been analysed.

Distribution of Animal Use Within Study Area

Between January 1995 and October 1998, 1,256 radiotelemetry locations were obtained from 43 individual radio-collared/implanted animals. Figure 4 depicts an adaptive kernal utilization distribution analysis of these locations. Four clear "peaks" in use (25% probability contour) are

Table 3. Home range analysis results for 10 male and 10 female wolverine using 100% Minimum Convex Polygon (MCP) method.

Males	Age class[a]	No. of locations	Home range (km^2)	Females	Age class[a]	No. of locations	Home range (km^2)
M202	A	81	689.5	F206	A	96	405.7
M203	A	59	875.4	F213	A	104	520.1
M208	A	50	568.9	F222	A	84	270.3
M214	A	67	369.8	F228	A	80	150.2
M224	A	30	754.8	F229	A	54	443.7
M225	S	39	874.3	F235	S	23	472.7
M226	S	22	1636.1	F236	A	39	217.9
M233	S	24	2884.3	F223	S	24	149.1
M237	S	25	1049.7	F241	S	17	204.3
M240	A	26	347.3	F242	S	31	271.2

[a] A = adult; S = subadult.

Table 4. Capture matrix and results for Jolly-Seber population estimate based on wolverine live-trapping data 1995–98.

	1995	1996	1997	1998
Captures	13	15	13	18
Recaptures				
1995		4	2	
1996			4	3
1997				6
Population estimate (95% CI)		25.6 (15.8–55.7)	24 (14.7–43.1)	

evident; the largest is located in Glacier National Park, followed by Mount Revelstoke National Park, Bigmouth/Windy Creek (north), and Downie/Goldstream (centre). Overlay techniques clearly demonstrate that high-use areas ("peaks") are found in protected areas in greater proportion than expected based on relative trapping effort and total park area. Protected areas comprise approximately 20% of the study area and include approximately 11% of the high trapping-effort area (25% contour), but contain >68% of the high-use area (Figs. 2 and 4).

Population Size

A Jolly-Seber mark–recapture estimate of population size using live-trapping data yielded estimates of 25.6 and 24.0 (Table 4) for the 4,000-km^2 core study area in 1996 and 1997 respectively. We did not correct for partial residency of animals (Garshelis 1992). Estimates suggest that approximately 50% of the wolverine population was marked during the census interval.

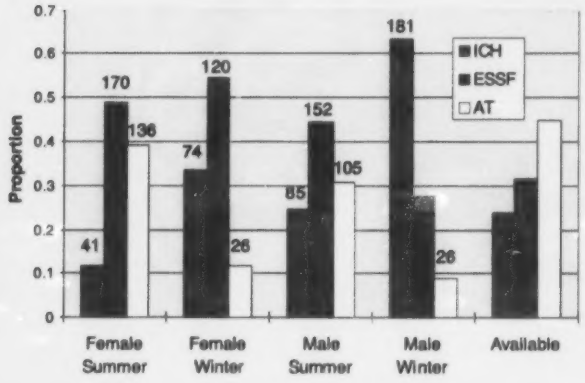

Figure 3. Seasonal habitat use of male and female wolverine by biogeoclimatic zone. (ICH = Interior Cedar–Hemlock; ESSF = Engelmann Spruce–Subalpine Fir; AT = Alpine Tundra.) Results were pooled among individuals for analysis.

DISCUSSION

Estimates of vital rates are essential to assess population growth or decrease (e.g., grizzly bears: Eberhardt et al.1994, Hovey and McLellan 1996, Mace and Waller 1997) and to aid in establishing sustainable harvests. To our knowledge, field estimates of wolverine survival are not available and reproductive rate data are scant. Banci (1994) summarized numbers of mortalities reported in radiotelemetry studies, but these were not reported as rates. Magoun (1985) suggested that annual survivorship would need to be ≤0.906 for a hypothetical wolverine population in NW Alaska to be stationary or stable. Our estimate of 0.77 falls well below this threshold, which may indicate a decline. Closer examination of the assumptions used in the Alaskan analysis will be required to determine if this is the case. In Idaho, Copeland (1996) documented a reproductive rate of 0.67 kits/female/year; Magoun (1985) in NW Alaska reported 0.69 kits/female/year. Our present estimate of 0.43 kits/female/year is lower, but is based on a low sample size. Differences arise through strong year effects, presumably related to food availability. In 1997 we had 2 litters from 5 adult females, whereas in 1998 only 1 litter was produced from 6 females. These rates are well below the averages reported using corpora lutea, placental scars, and fetus counts from carcasses (Rausch and Pearson 1972, Liskop et al. 1981, Banci and Harestad 1988). However, early survival of neonates may be low during periods of food stress. Lactation is known to be the most energetically demanding period for females. For wolverine this would correspond to the 15 February–30 April period. During years with poor carrion availability, reproductive females may fail to meet the high energetic demands of lactation and lose their litters (Magoun 1985). This could explain why some females in our study established what appeared to be natal dens but were never observed with kits after den abandonment. Since rates derived from carcasses do not capture the critical stage of preweaning kit survival, field estimates provide a better estimate of realized reproductive rate. At the conclusion of the project, reproduction and survival data will be used to explore potential rates of increase for our study population.

Reproductive females established dens in areas with little or no human disturbance, below treeline under avalanche debris or large boulders. Although at the stand or patch scale all dens located to date have been found in nonforested habitats similar to those reported by Magoun and Copeland (1998), our data differ when viewed at the landscape scale. Results clearly suggest that the upper-elevation forested zone (ESSFvc), not the alpine/parkland zone, is most used for denning. Additional factors, such as human activity, distribution of prey/carrion, and presence of other predators, likely affect the suitability of an area as denning habitat, but have not yet been investigated.

Carrion from avalanche-killed ungulates, other predator kills, and the presence of nutritionally stressed moose and goats likely explain the extensive use of valley-bottom ICH habitats by male wolverine during winter. Females also made greater use of the ICH in winter than summer; however, use of the ESSF zone was greatest. This may be partially an artifact of the location of reproductive dens. Copeland (1996) also recorded a seasonal shift in habitat use from montane forest types in winter to higher-elevation nonforested habitats in summer. Female wolverine in our study were frequently observed hunting and/or feeding on hoary marmot (*Marmota caligata*) during late spring and summer in alpine and subalpine habitats. Since wolverine operate at the landscape scale, with home ranges averaging 300 km^2 for females and 1,000 km^2 for males in our study area, habitat selection is unlikely to be strongly tied to stand or patch level attributes. Rather, habitat patch size, juxtaposition, and prey density/distribution may be more important factors. Final habitat use analysis will evaluate these factors in a GIS (geographic information system) environment.

Density estimates reported from published studies utilizing radiotelemetry and mean home range size tend to yield higher and more variable density estimates (1/65 km^2, Hornocker and Hash 1981; 1/48 km^2, Magoun 1985; 1/177 km^2, Banci 1987; 1/198 km^2, Copeland 1996) than those based on snow-tracking (1/207 km^2, Quick 1953; 1/193 km^2, Becker 1991). Whether these differences are related to the estimation techniques or actual differences in wolverine density is not certain. Our estimate (1/167 km^2) using 4 years of live-trapping information falls within the range of those reported above. A consistent problem in all the techniques employed is the relatively small study areas/high amount of edge, where estimates are derived that may inflate estimates significantly. This closure bias may artificially inflate estimates. The approach of Garshelis (1992), where telemetry data is used to adjust for animals who reside partly outside the census area, may provide a practical means for making more reliable estimates in small study areas; alternatively, larger census areas with less edge effect may be effective if used with track-based (Becker 1991), motion-sensitive cameras (Mace et al. 1994), or DNA-based methods (Woods et al. submitted) now being developed for grizzly bears.

The uneven distribution of wolverine use across the study area (Fig. 4) suggests that habitat quality is not equal across

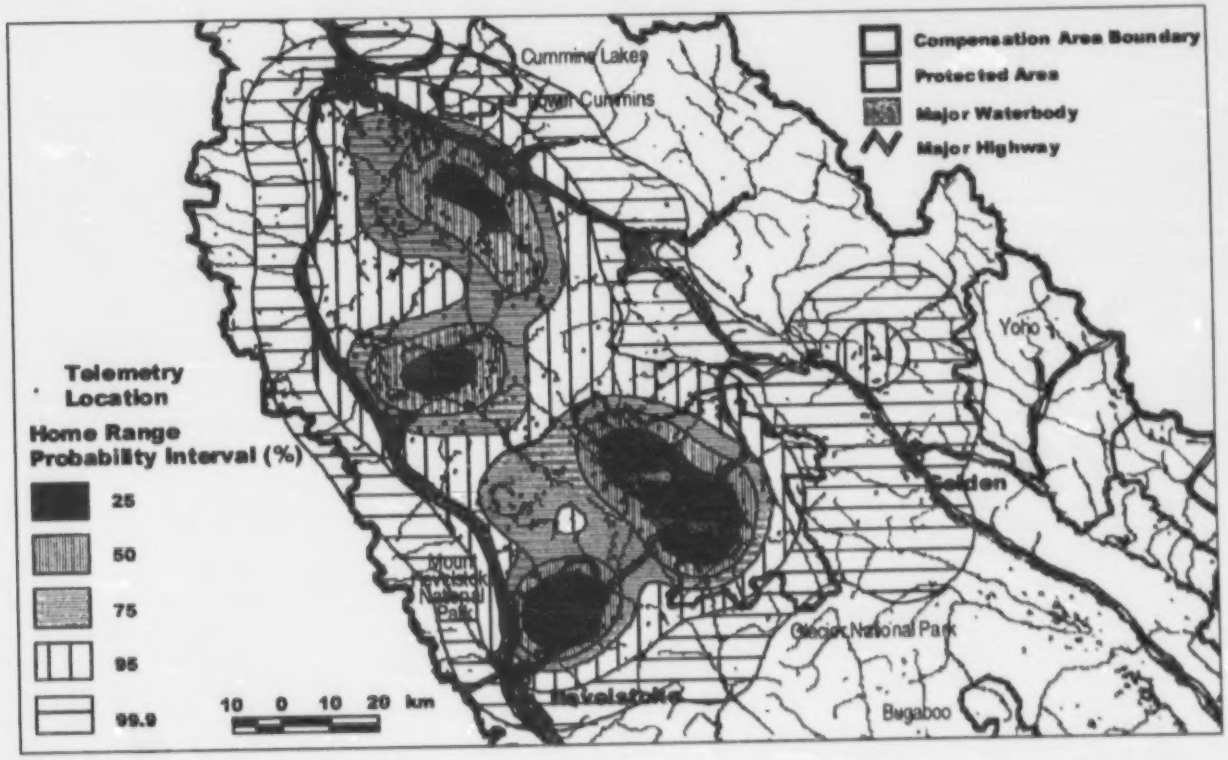

Figure 4. Adaptive kernal utilization distribution of all wolverine telemetry locations, 1995–January 1999. High-use areas show up as dark grey.

the area. The 2 largest "peaks" in use are centred on Glacier and Mount Revelstoke national parks. The Upper Bigmouth/Windy Creek "peak" area appears to support a high proportion of the wolverine population in the northern portion of the study area and may function as a refugium (Hatler 1989), similar to the national park areas. There is strong evidence that considerable movement occurs between Bigmouth Creek watershed and Windy and several smaller tributaries (Trident Creek; 2 unnamed creeks), which drain into Kinbasket reservoir. At present, most of this area is a defacto wilderness with access only in Bigmouth Creek. The high-use area located in the Downie/Goldstream area includes Nightmare Creek, 2 small Goldstream River tributaries, and the upper portions of Granite and Long creeks, which drain into Downie Creek.

MANAGEMENT IMPLICATIONS

Recommendations are preliminary at present, pending completion of field data collection.

1. Human-caused mortality of wolverine from trapping and transportation corridors is the largest factor influencing survivorship. Trapping restrictions may be warranted if rates are found to be unsustainable. Tracking sex and age of harvested animals through compulsory inspection would assist management decisions. Carrion along the road and rail right-of-ways needs to be disposed of rapidly to avoid collateral kill of carnivores such as wolverine.
2. National parks and unroaded wilderness areas appear to act as refugia at present. Pressures from commercial backcountry use, snowmobiling, and logging may erode the capacity of these areas to support wolverine, particularly reproductive females.
3. Maintenance of an abundant, diverse ungulate community is a necessary precursor to persistence of wolverine. Habitat and harvest management strategies that maintain moose, mountain goat, and caribou populations will benefit wolverine.

ACKNOWLEDGEMENTS

Special thanks to funding partners Columbia Basin Fish and Wildlife Compensation Program (CBFWCP), Habitat Conservation Trust Fund, Parks Canada, and British Columbia Forest Service for supporting this work, and to our families for putting up with our frequent absences. Numerous people have assisted on this project. D. Mair and J. Hooge collected superior aerial telemetry location data; J. Townley and C. Gillies assisted on snow trailing and trap maintenance; R. Klafki assisted with live-trapping, testing the motion sensitive cameras, and DNA Stations. J. Flaa, J. Woods, B. McLellan, F. Hovey, D. Adama, and L. Ingham pitched in to assist with various activities including live capture and handling, and maintaining the telemetry database. Thanks to I. Parfitt who created the GIS maps from the telemetry data, and completed the BEC zone area and location summaries within a tight time frame. Canadian Mountain Holidays provided logistical support from their lodges in the Goldstream River. Several park wardens assisted on field excursions from time to time throughout the life of the project. Discussions with E. Lofroth, G. Mowat, J. Copeland, and A. Magoun were helpful. D. Boles assisted with construction of traps, bait collection, and provided some lure at the outset of the project. CBFWCP summer students C. Gillies, R. Franken, and R. Klafki assisted with trap construction.

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Wolverine Winter Travel Routes and Response to Transportation Corridors in Kicking Horse Pass Between Yoho and Banff National Parks

M.A. Austin

Wildlife Branch, British Columbia Ministry of Environment, Lands and Parks
PO Box 9374 STN PROV GOVT, Victoria, BC, V8W 9M4, Canada
Matt.Austin@gems7.gov.bc.ca

S. Herrero

Department of Environmental Design, University of Calgary
2500 University Dr. N.W., Calgary, AB, T2N 1N4, Canada

P. Paquet

Department of Environmental Design, University of Calgary
2500 University Dr. N.W., Calgary, AB, T2N 1N4, Canada

ABSTRACT

During 2 winters we investigated the influence of transportation corridors on wolverine (*Gulo gulo*) movements through snow-tracking. Analysis of movements within the study area showed avoidance use of areas within 100 m of the Trans-Canada Highway and preference for areas >1100 m from the highway. Analysis of movements on the ski trail that formed the boundary of the study area showed a similar trend with avoidance of sections of the trail within 200 m of the highway and preference for sections >1100 m away. Mean width of the right-of-way for highway crossings by wolverines (68 m) was significantly shorter than that of approaches without crossing (165 m). Wolverines approaching the highway made repeated approaches and retreats and only crossed 3 out of 6 times. Wolverines did not hesitate to cross the Canadian Pacific Railway in areas where it does not share a common right-of-way with the highway. Our results indicate that Kicking Horse Pass is an important movement corridor for wolverines, however, the Trans-Canada Highway currently appears to be having a significant impact on wolverine movements. We believe that roads with narrow rights-of-way (<50 m) have less impact on wolverine movements than roads with wide rights-of-way (>100 m). We expect the impact of the highway on wolverines to be greater during the summer when traffic volume is higher.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Microsatellite Analysis of North American Wolverines

Christopher J. Kyle
Department of Biological Sciences, University of Alberta
Edmonton, AB, T6G 2E9, Canada
ckyle@ualberta.ca

Curtis Strobeck
Department of Biological Sciences, University of Alberta
Edmonton, AB, T6G 2E9, Canada

ABSTRACT

Elucidating the population genetic structure of a species gives us insight into the levels of gene flow and connectivity between geographic regions. These are important considerations when trying to manage a threatened or Blue-listed species, such as wolverines, as we are then provided with information on how far and where animals may be dispersing and where population boundaries occur. Population genetic structure can be estimated using fast evolving, highly variable molecular markers such as microsatellites. These molecular markers have a high mutation rate, can be analyzed from a very small amount of DNA using polymerase chain reaction, and provide reproducible results. Using microsatellites, we have addressed questions of genetic variation within populations with estimates of population heterozygosity, mean number of alleles, and probability of identity. Gene flow among populations was also estimated via an assignment test and genetic distance measures. In this preliminary study, 4 geographic regions (2 from the Northwest Territories and 2 from British Columbia) were sampled and analyzed at 7 microsatellite loci. As expected, more gene flow occurs between proximal regions than those separated by large geographic distances. Our preliminary results also suggest that there is a directional trend of gene flow from the Northwest Territories and Revelstoke to northern British Columbia, whereas the Revelstoke region seems to be more genetically isolated from the other regions analyzed here.

Key words: gene flow, genetic structure, *Gulo gulo*, microsatellites, population, wolverine.

The population genetic structure of North American wolverines (*Gulo gulo*) is unknown to date. With increasing concern about the status of this forest carnivore and how it might be managed effectively, this information is needed to determine what constitutes a population of wolverines and what levels of effective migration occur between these geographic areas. Elucidating the population genetic structure of a species can also provide estimates of a species' vagility, which can then be correlated to ecological estimates of dispersal. Furthermore, successful reintroduction programs, ideally, reintroduce animals that come from a similar genetic background to those that once existed in that area. Hence, determining the population genetic structure may help to establish which populations are best suited for reintroduction into a depauperate area.

Molecular markers, such as microsatellites, can provide estimates of genetic distance between populations which reflect the effective migration between populations, and not simply movement between them. Direct observations of individuals and migration rates between separate geographic areas are often very difficult and costly, if not impossible for some species when attempting to estimate gene flow. Furthermore, direct observations of migration can be misleading if migrants do not contribute to the gene pool of the population they have moved into. When the population genetic structure is related to the levels of gene flow between geographic areas it enables us to determine how, and if, individuals are moving between populations. From this we can also determine population boundaries for a particular species (if they exist). Populations can be identified by the genetic distinctiveness of individuals taken from different geographic areas which may not always be evident from direct observations.

Measures of migration and population boundaries are important factors in managing a species. First, we can use them to elucidate whether a management unit is a distinct population or simply part of a larger panmictic population. Second, we can determine if a harvested population is being replaced solely from within by the level of fecundity in that population or if replacement from other populations plays a role in its numbers. Levels of genetic variation within populations can also help identify which populations (if any) act as source populations for smaller and more isolated regions. This becomes an important factor when a species' habitat becomes fragmented, separating source populations from sink populations. When this occurs, concerns for inbreeding may arise.

Gene flow, and hence the population genetic structure of a species, are determined by the life history characteristics of that species. Those animals with larger home ranges and elevated levels of vagility are likely to have less population

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

genetic structure that those with smaller home ranges and limited vagility. Other life history traits such as fecundity, longevity, and density are also important in determining population genetic structure.

Wolverines have relatively large home ranges and can disperse across large distances in a short time, especially as juveniles (Gardner 1985). Furthermore, *Gulo gulo* is thought be relatively diffuse throughout its distribution (Banci 1994), especially in the extreme southern regions of this animal's distribution. These characteristics of high vagility, large home ranges, and relatively high longevity might be expected to lower the levels of population subdivision in this species. Yet, low population numbers in the southern regions of this species' distribution and high levels of juvenile mortality as they disperse may lead to very isolated populations. If genetic variation does exist among populations, high resolution (fast evolving) neutral genetic markers will be needed to detect it. For this reason we have chosen to use hypervariable polymorphic microsatellites as in other mammalian studies of species with high vagility, such as the polar bear (*Ursus maritimus*). Clear genetic differentiation was found in these animals using microsatellites despite the known long range movements of the species and that little variation was detected using other methods (Paetkau et al. 1995).

STUDY AREAS AND METHODS

Seven polymorphic loci were used to examine the population genetic structure of *Gulo gulo* from 4 geographic regions including: the Kugluktuk (N = 45) and Bay Chimo (N = 40) regions in the Northwest Territories (NWT) and the Revelstoke (N = 48) and Williston Lake (northern B.C.; N = 37) regions in British Columbia (Fig. 1). Samples were provided as frozen tissue, ear plugs, and hair roots. DNA was extracted using a QIAamp® Tissue Extraction Kit. The 7 loci used to amplify the microsatellites were developed by Davis and Strobeck (1998) in badgers (BA-1 and BA-4) and wolverines (GG-3, GG-4, GG-7) and by Duffy et al. (1998) in wolverines (Ggu 101 and Ggu 234). DNA at all loci was amplified using the same protocol as in Davis and Strobeck (1998). The DNA fragments were then visualized using an ABI Prism™ 377 DNA sequencer. DNA fragments were analyzed using the programs GeneScan™ Analysis 2.02 and Genotyper® 2.0.

Figure 1. Map of sampled regions. Dot size reflects relative size of sampled regions.

Data Analysis

To assess the relative genetic variation of each of the 4 geographic areas sampled the allele frequencies for each loci were first determined. From these frequencies the mean number of alleles, average heterozygosity, and overall unbiased probabilities of identity (Paetkau et al. 1998) were calculated. The genetic distances among the 4 geographic areas were estimated using 2 measures: 1) both Nei's standard genetic distance, D_S, which is calculated from genotype frequencies (Nei 1972); and 2) the genotype likelihood ratio, D_{LR}, calculated from genotype probabilities (Paetkau et al. 1997). An unrooted tree of the D_{LR} values was then created using PHYLIP 3.572 (Felsenstein 1995). The assignment test, which determines the probability of a genotype occurring in the region from which it was sampled and the probabilities of it occurring in the regions it is being compared with (Paetkau et al. 1995), was run for all 4 geographic areas. The programs used to calculate the assignments and the genetic distance measure (D_{LR}) were designed by J. Brzustowski and are available at:
http://www.biology.ualberta.ca/jbrzusto/alpha/Doh.html.

The assignment test (Paetkau et al. 1995), also found on the website by J. Brzustowski, was run for all populations. This program determines both the probability of a genotype occurring in the region from which it was sampled and the probability of it occurring in the regions it is being compared with. It then assigns individuals to the population for which that individuals genotype has the highest probability of occurring. The assignment test was run several times using the following options: replacement of gene frequency values of 0 with 0.01, resampling of the populations from within each gene pool assuming Hardy-Weinburg equilibrium (HWE), and resampling of the populations by combining the gene pools. The option of resampling the data from each gene pool lends a significance value to the assignment of individuals to particular populations, and may help identify potential migrants between populations. The randomizations of the data indicate the percentage (of the number of iterations used) of when more individuals were assigned to a population than the original assignment. Therefore, if a small percentage is obtained from the randomization then it is unlikely that all of the misassignments of individuals are due simply to

Table 1. Sample size, mean number of alleles, mean heterozygosity, and overall unbiased probability of identity for each of the 4 geographic sampling regions.

Population	N	Mean no. of alleles	Mean heterozygosity	Prob. of identity[a]
Revelstoke	48	4.27	57.92	1/42,407
Northern B.C.	37	4.57	61.79	1/254,373
Bay Chimo	40	4.71	62.14	1/157,299
Kugluktuk	45	4.71	63.99	1/302,836

[a] $\sum_i p_i^4 + \sum_i \sum_{j>i} (2p_i p_j)^2$

chance; some of the misassigned individuals are most likely migrants from the population to which they were misassigned into the one from which they were sampled. When the option of combining the gene pools is used, the value obtained suggests how heterogeneous the populations are. If most of the randomizations of the populations sets have fewer self-assignments from a population to itself, then this is evidence for population structure and not simply a reflection of regional sampling efforts.

We chose the following options: 1) replacement of allele frequency with a value of 0.01 instead of 0; 2) resampling (10,000 times) of the populations from within each gene pool assuming HWE; and 3) resampling (10,000 times) of the populations by combining the gene pools. The second option lends a significance to the individual assignments. The third option indicates that misassignments are due to "true" migration between the 2 regions compared, or simply that the probability of the genotype occurring in either region is low. A high value from the resampling results for the individuals correctly assigned implies that any individuals misassigned are truly migrants between those two regions. Conversely, a low value indicates that the origin of the misassigned individuals has no significance.

RESULTS

Four geographic areas were sampled in this preliminary study (Fig. 1). The relative measures of genetic variation among the sampled regions are summarized in Table 1. For all 3 measures (mean number of alleles, mean heterozygosity, and overall unbiased probability of identity), the Revelstoke region displayed the least variation, whereas the Kugluktuk region displayed the most. The mean number of alleles (ranging from 4.57 to 4.71), and the mean heterozygosity (ranging from 57.92 to 63.99%) did not vary greatly amongst these 4 regions. The unbiased probabilities of identity estimates, however, did have a greater range with values from Revelstoke having a probability of 1/42,407 to Kugluktuk having a value of 1/302,836.

Two estimates of genetic distance were calculated, D_{LR} and D_s (Table 2). The estimates show that Revelstoke is genetically more similar to the northern B.C. area (D_{LR} = 0.544, D_s = 0.0565) than the Bay Chimo or Kugluktuk areas from which it is roughly generically equidistant (D_{LR} = 0.922, D_s = 0.0806 and D_{LR} = 1.09, D_s = 0.0922, respectively). The northern B.C. area is roughly genetically equidistant from all 3 other areas with values of D_{LR} ranging from 0.534 to 0.582 and D_s ranging from 0.0422 to 0.0661 The 2 NWT areas are the mostly closely associated areas with a D_{LR} value of 0.240. An unrooted tree of the D_{LR} values (Fig. 2) reveals the relationship of the aforementioned genetic distances represented by the length of the tree branches.

The randomizations of the assignment test were run in 2 ways: 1) by resampling the regions from each gene pool assuming HWE, and 2) by combining all the gene pools assuming HWE (Table 3). The Revelstoke area had the fewest individuals assigned to other regions with 37/48 individuals assigned to itself and 5, 4, and 2 assigned to northern B.C., Bay Chimo, and Kugluktuk, respectively. The northern B.C. region had 14/38 individuals assigned to itself with 10 assigned to Revelstoke, 7 assigned to Bay Chimo, and 6 assigned

Table 2. Genetic distance estimates.

D_{LR}	Revelstoke	Northern B.C.	Bay Chimo	Kugluktuk
Revelstoke	0			
Northern B.C.	0.5440033	0		
Bay Chimo	0.9218481	0.58198035	0	
Kugluktuk	1.0903602	0.53401995	0.23970568	0

Nei's D_s	Revelstoke	Northern B.C.	Bay Chimo	Kugluktuk
Revelstoke	0			
Northern B.C.	0.05646155	0		
Bay Chimo	0.08063514	0.06607150	0	
Kugluktuk	0.09221709	0.04246151	0.04215205	0

Table 3. Summary of assignment test results and randomizations of genotype frequencies.

Individual assignments

From:	To: Revelstoke	Northern B.C.	Bay Chimo	Kugluktuk
Revelstoke	37	5	4	2
Northern B.C.	10	14	7	6
Bay Chimo	2	7	20	11
Kugluktuk	4	3	14	24

Randomizations from within each gene pool

	Revelstoke	Northern B.C.	Bay Chimo	Kugluktuk
Revelstoke	3,154	7,045	6,854	7,503
Northern B.C.	1,050	9,970	1,284	2,350
Bay Chimo	9,488	986	9,252	1,228
Kugluktuk	5,528	9,061	387	8,671

Randomizations from the combination of all gene pools

	Revelstoke	Northern B.C.	Bay Chimo	Kugluktuk
Revelstoke	0	9,972	9,992	10,000
Northern B.C.	945	4,510	8,603	9,346
Bay Chimo	9,997	9,087	27	4,161
Kugluktuk	9,985	9,997	2,175	6

to Kugluktuk. Bay Chimo had 20/40 individuals assigned to itself, 2 to Revelstoke, 7 to northern B.C., and 11 to Kugluktuk. The Kugluktuk region had 24 animals assigned to itself, 4 to Revelstoke, 3 to northern B.C., and 14 to Bay Chimo.

CONCLUSIONS

The relative measures of genetic variation (mean number of alleles, mean heterozygosity, and unbiased probability of identity) among the geographic areas sampled suggest that less variation is found from northern to southern regions

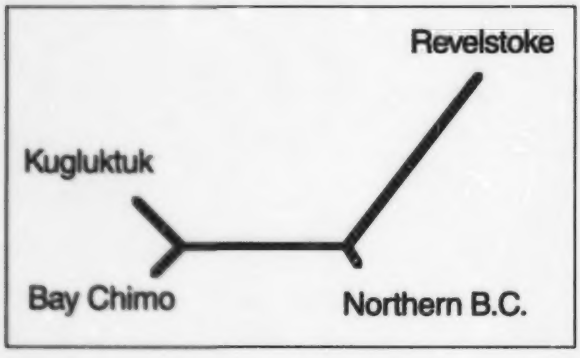

Figure. 2. Unrooted neighbour joining tree of genotype likelihood ratio values.

Table 4. Straight line distances (km) between centres of sampled regions and their approximate areas.

	Revelstoke	Northern B.C.	Bay Chimo	Kugluktuk
Revelstoke (7,000 km²)	0			
Northern B.C. (10,000 km²)	550	0		
Bay Chimo (15,000 km²)	1,550	1,200	0	
Kugluktuk (15,000 km²)	1,450	1,000	350	0

(Table 1). The relative size of each sampled area, however, could be influencing these results (Table 4). The Revelstoke area has the least variation, but it is also the smallest region sampled (approx 7,000 km^2), which may account for the lower mean number of alleles, mean heterozygosity, and unbiased probability of identity estimates. More likely, however, is that gene flow towards Revelstoke from the other areas sampled is somehow impeded, which accounts for its decreased genetic variation.

The 2 genetic distance measures, D_{LR} and D_S, present similar results (Table 2). Both measures suggest that the northern B.C. area is roughly genetically equidistant from both the Revelstoke area (D_{LR} = 0.544, D_S = 0.0565) and the NWT regions of Bay Chimo (D_{LR} = 0.582, D_S = 0.0661) and Kugluktuk (D_{LR} = 0.534, D_S = 0.0425). These results also suggest that the 2 NWT regions are relatively close genetically (D_{LR} = 0.240, D_S = 0.0421). The more proximal regions (Fig. 2) are more closely related genetically. The Revelstoke region, however, is disproportionately genetically distant from the northern B.C. region as compared with the NWT regions when taking their geographic locations into account (Tables 2 and 4). Once again, this result suggests that some factor impedes gene flow between the Revelstoke area and the other regions sampled, and there are fewer impediments among the remaining 3 areas.

The randomization of the genotype frequencies in the assignment test (Table 3), with all gene pools combined, suggests that there is a significant level (95%) of population genetic structure between all of these areas except the northern B.C. region. This region revealed a slightly less significant level of structure (90% confidence). The test also estimated that 37/48 animals were correctly assigned to Revelstoke from Revelstoke, whereas only 14/37 in northern B.C., 20/40 in Bay Chimo, and 24/45 in Kugluktuk were correctly assigned (Table 3). These results, when compared with one another, suggest that some impediment to gene flow into Revelstoke from the other regions exists. Note that these assignments do not necessarily mean that individuals captured in one region and assigned to another are actually migrants from one population to the other. What it does

demonstrate is that the probability of a particular genotype occurring in an area with similar genotypes is higher.

To estimate the levels of migration among the sampled groups the assignment test was run using the option of resampling the genotypes from their original gene pools (Table 3). The results for Revelstoke region suggest that there is little migration into the Revelstoke region; the number of misassigned individuals into this population is not well supported by the resampling of the genotype frequencies, whereas movement from Revelstoke to northern B.C. is well supported by the randomizations. The northern B.C. region seems to be receiving the most gene flow into it from both Revelstoke and the NWT regions. Ten individuals sampled from northern B.C. are more likely to have come from the Revelstoke region, while 7 and 6 individuals are more likely to have come from Bay Chimo and Kugluktuk, respectively. The resampling of these genotypes supports (approx 90% significance for individuals from Revelstoke and Bay Chimo) the initial assignment of individuals to each region, and that these individuals are actually migrants into the northern B.C. region. The Bay Chimo and Kugluktuk regions reflect the highest levels of gene flow between 2 groups with 14/45 individuals going from Bay Chimo to Kugluktuk and 11/40 going from Kugluktuk to Bay Chimo. This is expected from their geographic proximity. The gene flow to these NWT regions from Revelstoke is not significant. The gene flow from northern B.C. is a bit higher, however, with 7 individuals assigned (with 90% significance) to northern B.C. that were sampled in Bay Chimo.

The assimilation of the 3 types of data analyzed in this study (genetic variation, genetic distance, and gene flow via the assignment test) suggests that there is some impediment to gene flow into the Revelstoke region. The northern B.C. region, having less genetic structure and receiving the most gene flow into it from both northern and southern regions, seems to be acting as a sink population. However, only 4 regions have been analyzed in this preliminary study. The 2 NWT regions, although displaying some population genetic structure, do have extensive gene flow between them. This structure was not expected because of the ability of these animals to disperse and the more contiguous landscape in the northern regions of NWT. Further investigation with more loci and more sampled regions will enable us to refine and resolve the suggested levels of gene flow found in this species across its North American distribution.

ACKNOWLEDGEMENTS

We thank the following people for providing samples for this study: J. Krebs and D. Lewis from the Columbia Basin Fish and Wildlife Compensation Program; D. Reid and E. Lofroth from the Northern Wolverine Project; and R. Mulders and A. Bourque from the Department of Resources, Wildlife and Economic Development, NWT. This research was funded by Natural Sciences and Engineering Research Council and Parks Canada grants to C. Strobeck.

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Lynx Ecology in the Southern Canadian Rocky Mountains: Preliminary Results and Conservation Implications

Clayton D. Apps
Aspen Wildlife Research
2331 - 7th Avenue N.W., Calgary, AB, T2N 1A1, Canada
aspen@cadvision.com

Alan Dibb
Parks Canada
P.O. Box 220, Radium Hot Springs, BC, V0A 1M0, Canada

Anna J. Fontana
British Columbia Ministry of Environment, Lands and Parks
205 Industrial Road G, Cranbrook, BC, V1C 6H3, Canada

ABSTRACT

Throughout the range of the Canada lynx, snowshoe hares are considered their primary prey. Relative to northern populations, hares occurring in mountainous regions at southern latitudes are thought to remain at low and stable densities. Hence, the ecology of southern lynx populations is expected to resemble that of northern populations during the low phase of the hare population cycle. The space use, diet, and demographics of lynx in the Rocky Mountains of southern British Columbia and Alberta are consistent with this hypothesis, based on data collected from 10 lynx, including 6 (3M, 3F) resident adults, during 2 years of an assumed increase phase of a hare cycle. Mean hare densities were low, ranging from 0.01–0.47/ha among cover types and landscapes. Lynx diet (n = 137 kills) was diverse, and included hares (52%), red squirrels (30%), northern flying squirrels (5%), grouse (3%), martens (3%), and voles (3%). Kitten recruitment to winter was 0 among adult females for 4 lynx-years. Family groups that did occur in the study area during winter were associated with small litters of 2. Survival among resident adults was 100%, but 3 of 4 subadults monitored during winter did not survive to mid-May. Home ranges were large, with annual 95% adaptive kernel utilization distributions averaging 381 and 239 km2 for resident males and females respectively. Minimum daily movements averaged 3.8 and 3.0 km respectively. Two juvenile dispersals were short (44 and 17 km) and ended in starvation. Results suggest that lynx populations in the southern Canadian Rocky Mountains are not subject to the dramatic cyclic pulses in productivity reported for northern populations. Coupled with evidence that resident animals are patchy in distribution, this implies that lynx are vulnerable to mismanagement of habitat and populations within this region.

Key words: Canada lynx, demographics, diet, dispersal, home range, *Lepus americanus*, *Lynx canadensis*, movement, recruitment, Rocky Mountains, survival, snowshoe hare.

There is concern for the conservation of Canada lynx (*Lynx canadensis*) populations occurring near the species' southern range extents, as has recently been highlighted by a proposal for listing under the federal Endangered Species Act within the conterminous United States (Fish and Wildlife Service 1998). However, the status of southern lynx populations, and associated threats, are clouded by a lack of information regarding their local ecology, demographic trends, and habitat associations. Most research has occurred within northern boreal forests more central to the species' geographic range. In comparison, regional landscapes with which lynx are associated in southern British Columbia and Alberta and the northwestern Unites States are typically mountainous and vary greatly in their inherent potential to support lynx. Capable habitat is therefore limited in area and naturally fragmented relative to northern landscapes, and is subject to multiple demands of a growing human population.

Ecotypic differences may also occur between northern and southern lynx populations, important considerations for population and habitat management. Lynx ecology is closely tied to the density and distribution of snowshoe hares (*Lepus americanus*), their primary prey. In the north, hare densities vary dramatically over an approximate 10-year cycle, influencing lynx demographics (Brand et al. 1976, Brand and Keith 1979, Poole 1994, Mowat et al. 1996, Slough and Mowat 1996, O'Donoghue et al. 1997), food habits and foraging behaviour (Brand et al. 1976, O'Donoghue et al. 1998*a*,*b*), space

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

use and movements (Ward and Krebs 1985, Poole 1994, Slough and Mowat 1996), and dispersal (Poole 1997). However, it appears that hare populations of southern latitudes and mountainous regions remain at relatively low and stable densities through time, possibly as a result of a more patchy habitat distribution, greater competition, and a greater suite of predators (Dolbeer and Clark 1975, Wolff 1980). Hence, Koehler (1990) theorized that the ecology of southern lynx populations resembles that of northern populations during cyclic hare lows. Assuming that any variation of hare density will be synchronous throughout their range (Smith and Davis 1981), it follows that several hypotheses would hold true during the increase and high phase of the cycle. Relative to results reported for northern populations during hare lows, southern lynx populations should be associated with: 1) hare densities that are as low; 2) a diet of as much alternate prey; 3) resident home ranges that are as large; 4) daily foraging movements that are as long; 5) reproduction and survival that are as low; and 6) dispersal rates that are as great.

Space use by lynx may also relate to physiography (Koehler and Aubry 1994), the influence of which may vary seasonally in mountainous landscapes. For example, open habitats that occur in conjunction with higher elevations and rugged topography are associated with several prey species that would be available only during snow-free months (Scotter and Ulrich 1995). Snow conditions that may restrict lynx movements and/or increase expended energy (Murray and Boutin 1991) are also more likely at higher elevations. Thus, individuals occurring in mountainous terrain can be expected to make shorter daily movements and use smaller home ranges when snow is present.

We examine the above hypotheses with preliminary results from an ongoing study in the southern Canadian Rocky Mountains. Data were collected between November 1996 and October 1998, and are of an unexploited, southern lynx population, occurring in a highly mountainous region during the increase phase of a snowshoe hare cycle in other regions (C. Krebs, University of British Columbia, pers. comm.).

STUDY AREA

The study area (>3000 km2) is centrally located within the Shining Mountains ecoprovince (Demarchi 1994), and most of it lies within the East Kootenay Wildlife Management Subregion of southeastern British Columbia (Fig. 1). It is defined by the Beaverfoot and upper Kootenay drainages on provincial land, and straddles the Continental Divide of British Columbia and Alberta to include the Vermilion, middle Bow, and Kicking Horse valleys within Kootenay, Banff, and Yoho national parks.

Characteristic of the Canadian Rockies, much of the area is rock, ice, and other inherently unsuitable lynx habitat, and high peaks separate broad valleys incised by narrow tributary valleys. Elevations (1,200–>3,000 m) span 3 biogeoclimatic zones (Meidinger and Pojar 1991). Below 1,500 m, the Montane Spruce zone is characterized by a climax overstory of hybrid Engelmann/white spruce (*Picea engelmannii x glauca*) and subalpine fir (*Abies lasiocarpa*). The Engelmann Spruce–Subalpine Fir zone occurs at higher elevations but below 2,300 m, and the Alpine Tundra zone occurs at the highest elevations. Subclimax stands throughout the study area are dominated by lodgepole pine (*Pinus contorta*). The macroclimate is continental, with mean temperatures ranging from −18°C (January) to 23°C (July), and 42–63% of the 51–81 cm mean annual precipitation falling as snow (Achuff et al. 1984).

Several other predators and potential competitors with lynx occur in the study area (Scotter and Ulrich 1995). Potential lynx prey include snowshoe hares, red squirrels (*Tamiasciurus hudsonicus*), northern flying squirrels (*Glaucomys sabrinus*), gallinaceous birds, microtine rodents, hoary marmots (*Marmota caligata*), ground squirrels (*Spermophilus* spp.), bushy-tailed woodrats (*Neotoma cinerea*), beavers (*Castor canadensis*), and several ungulate species.

METHODS

We determined a range of snowshoe hare densities by count-

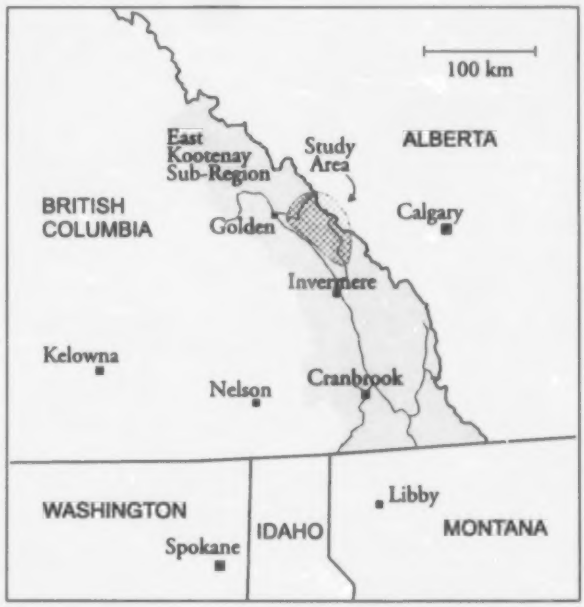

Figure 1. Study area location in the southern Canadian Rocky Mountains of southeast British Columbia and southwest Alberta.

ing fecal pellets during September 1998 in established 305 X 5.1 cm quadrats (Krebs et al. 1987). Ten quadrats were spaced at 30.5-m intervals along each of 61 transects. Transects were placed randomly among early-successional (20–60 yr old), mid-successional (60–120 yr old), and late-successional (>120 yr old) stands, but were spaced by 500 m. Sampling was further stratified among 3 landscapes within the study area, representing geographically distinct areas where we consider terrain conditions conducive to supporting lynx. Hare densities were calculated with a modified version of Krebs et al.'s (1987) regression equation, using the HARETURD program (C. Krebs, University of British Columbia, 27 Oct 1998, pers. comm.).

Study animals were captured during the winters of 1996–97 and 1997–98, with efforts focused in landscapes where lynx sign had been reported in recent years. Capture, handling, and radiotelemetry methods are detailed by Apps (1999). Upon capture, lynx were sexed, and age was subjectively classed (adult: >2 yr; juvenile: 1–2 yr; kitten: <1 yr) based on skeletal measurements and tooth wear. They were then fitted with motion/mortality-sensing whip antennae radio-collars (Lotek Eng. Inc., Newmarket, ON) before release. Radio-locations were obtained throughout the year, at least weekly for all animals, and at least 4 days per week for most animals.

Study animals were snow-tracked during both years when possible, and kills and scavenging were documented over 759 km. Prey remains were sufficient for positive species identification in most cases, although some kills may have been missed due to recent snow or for small prey. Tracks were directly observed wherever possible to minimize missed kills.

Lynx carcasses and/or radio-collars were retrieved as soon as possible after mortality was detected. Probable cause of death was determined from necropsies (Shury Veterinary Services, Canmore, AB) and field evidence. Female study animals were snow-tracked pre- and post-capture to determine whether kittens were present. When encountered, family groups of uncollared lynx were also snow-tracked to count kittens. The movements of females exhibiting strong site fidelity during May through July were radio-tracked daily.

We defined resident lynx as those associated with 1 activity centre subjectively determined over at least 3 months. Home ranges of resident adults were estimated seasonally and annually over both years using the program CALHOME (Kie et al. 1996). We defined summer and winter seasons by the typical 6-month snow and snow-free periods beginning and ending 1 November and 1 May. The late-February to early-April breeding period (Koehler and Aubry 1994) occurred in winter. To facilitate comparison with other research, we calculated the 100% and 95% minimum convex polygon (MCP; Hayne 1949), and 95%, 75% and 55% adaptive kernel utilization distributions (UD) using optimal smoothing parameters (Worton 1989). Lynx movements indicated that home ranges could be traversed within 24 hr, thus only radio-locations that were temporally independent by this interval were used (Swihart and Slade 1985). Among all animals, the 95% UD area appeared to asymptote at 43–55 radio-locations (Apps unpubl. data). We compared UD levels between seasons using multivariate analysis of variance (MANOVA; $\alpha = 0.05$). Static interaction between lynx known to have neighbouring or contiguous home ranges was inferred from spatial home range overlap (Macdonald et al. 1980).

We defined minimum daily movements (MDM) as the distance between sequential radio-locations separated by 18–36 hr ($\bar{\chi}$ = 24 h). Mean MDMs were compared among resident lynx of each sex using analysis of variance (ANOVA; $\alpha = 0.05$), and between seasons using unpaired Student's *t*-tests ($\alpha = 0.025$). We assumed that lynx exhibiting consistent, linear movements were transient or dispersing animals. Where possible, the movements of these animals were summarized with respect to timing of dispersal, direction, distance, MDM, and proximate outcome.

RESULTS

Hare Densities

For early-, mid-, and late-successional stands respectively, mean hare densities by landscape were 0.16/ha, 0.08/ha, and 0.01/ha in the upper Kootenay Valley; 0.25/ha, 0.06/ha, and 0.10/ha in the Beaverfoot Valley; and 0.47/ha, 0.39/ha, and 0.32/ha in the Vermilion Pass.

Food Habits and Population Attributes

Ten lynx, comprising 3 adult males, 3 adult females, 1 juvenile male, 2 kitten/juvenile females, and 1 kitten male, were radio-collared, monitored, and snow-tracked for 1–23 months between November 1996 and October 1998. Lynx kills (n = 137) documented over both years consisted of snowshoe hares (52%), red squirrels (30%), northern flying squirrels (5%), grouse (*Dendragapus* spp.; 3%), martens (3%), voles (3%), a northern flicker (*Colaptes auratus*; <1%), and unknown species (3%). One case of scavenging on an intact mule deer, presumed to have died from a vehicle collision, occurred for 2–4 days before the lynx (M/01) was forced out by wolves and did not return.

None of the 3 adult females in the study sample were travelling with kittens at the time of their capture in November 1997 or 1998. One uncollared family group was detected in the study area during March 1997, and 3 uncollared family groups were detected during November and December 1997. We assumed the latter groups were different individuals because each group was separated by a linear distance of >30 km, beyond the maximum exploratory movement of radio-collared females (Apps unpubl. data). Each group contained an adult and 2 kittens. Both kittens (M/04, F/03) from 1 family group were subsequently radio-collared, but neither

Table 1. Annual minimum convex polygon (MCP) and adaptive kernel utilization distribution (UD) home range estimates (km2) for resident lynx in the southern Canadian Rocky Mountains, British Columbia and Alberta, 1996–98.

Lynx (sex/identity #)	Period monitored	n	MCP	UD			
			100%	95%	95%	75%	55%
M/01	11/96 – 10/98	448	559	357	337	117	45
M/02	03/97 – 10/98	189	346	224	330	49	24
M/03	03/97 – 06/98	141	388	249	477	150	51
Male mean		3	431	277	381	105	40
Male SD			113	71	83	52	14
F/01	11/96 – 09/97	200	408	276	505	85	31
F/02	11/97 – 10/98	254	133	85	94	25	11
F/04	04/98 – 10/98	103	77	44	87	30	13
Female mean		3	206	135	229	47	18
Female SD			177	124	239	33	11

survived to May. Two females that were monitored through March of 1997 (F/01) and 1998 (F/02) were snow-tracked while travelling with males, and F/02 was also directly observed during copulation with an uncollared male on 13 March. Daily radio-locations of F/01 were within 250 m of an activity centre between 20 May and 1 June 1997, after which she resumed normal movements and did not revisit the presumed den site. From this, we inferred that she lost or abandoned her litter at or just after parturition. Kittens were also not detected with her during snow-tracking early in the following winter. Radio-locations of F/02 were within 500 m of an activity centre on 82% of days between 22 May and 14 July 1998, and she was observed with 1 kitten in August of that year.

Known survival of resident adults was 100% over both years. A dispersing female kitten (F/03) and a transient, juvenile male (M/05) died of starvation on 15 April and 3 May 1998 respectively. A male kitten (M/04) likely was killed by an uncollared adult male lynx in December 1997 while travelling with his family group and within his natal range. A female kitten (F/05) released to her natal area in April 1998 after 4 months in captivity due to a capture injury survived at least the following 6 months.

Space Use and Movements

The mean number of radio-locations used for summer and winter home range estimates was 96 (range = 63–183) and 149 (range = 72–265) respectively. Mean 95%, 75%, and 55% UD estimates (Table 1) did not significantly differ between seasons for males ($F = 3.66$, 3 df, $P = 0.222$) or females ($F = 1.34$, 3 df, $P = 0.548$). Spatial overlaps between the annual 95%, 75%, and 55% UD areas of M/01 and F/01 were 36%, 63%, and 60%, respectively. Of their 183 concurrent daily radio-locations, 78% were separated by >3 km. A second adult female was known to occur also within the 50% UD area of these 2 lynx, as inferred from snow-tracking and radio-locations of her dependent kitten (M/04). In addition, unmarked adult lynx were detected within the 70% UD of all other study animals between 1 November and 15 February 1998. Minimum daily movements were 3.8 km ($n = 537$, SD = 3.5) and 3.0 km ($n = 404$, SD = 2.4) for males and females respectively. MDMs were greater during summer for M/01 ($t = 3.48$, 278 df, $P < 0.001$), and were greater during winter for F/02 ($t = 2.72$, 125 df, $P = 0.007$), while seasonal differences were not significant among other residents ($t < 1.77$, 26–110 df, $P > 0.082$).

Dispersal

Dispersal movements were documented for 3 subadult lynx. After he was radio-collared on 19 March 1998, a juvenile male (M/05) made a 44-km southeast movement over 11 days (MDM: $\bar{\chi}$ = 3.7 km, SD = 1.9). He then used 1 meadow complex intensively (MDM: $\bar{\chi}$ = 1.3 km, SD = 0.9) for the next 31 days before his death on 3 May. After she was radio-collared within her natal range, a female kitten (F/03) rejoined her mother for 3 days, then moved independently for 3 days before 28 March, when she initiated a southeast linear movement of 17 km over 3 days (MDM: $\bar{\chi}$ = 5.6 km, SD = 0.4). She then used 1 area intensively (MDM: $\bar{\chi}$ = 0.9 km, SD = 0.5) for the next 15 days before her death. The movements of both animals followed the major valley in which they occurred and paralleled but did not cross the Trans-Canada Highway. After her 30 April release, a kitten/juvenile female (F/05) used a 75% UD home range of 53 km2 for 5 months, after which she made 2 separate exploratory movements of a minimum 74 km and 55 km over 24 and 11 days respectively. The second movement continued in a southward dispersal.

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999.

DISCUSSION

Hare Densities

Hare densities (range = 0.01–0.47/ha) observed in the southern Canadian Rockies during 1996–98 are comparable to those reported in northern lynx study areas during cyclic hare population lows. Hare densities in the southwest Yukon declined from 8.0–10.7/ha to 0.2–0.5/ha during 1 cycle (Ward and Krebs 1985), and another population fell from 7.5/ha to 1.3/ha during the next cycle (Slough and Mowat 1996). Similarly, a population in the Northwest Territories fell from 7–9/ha to 0.4–1.0/ha during the early 1990s (Poole 1994), and a northern Alberta population crashed from about 17/ha to 0.34/ha during the early 1970s (Brand et al. 1976). Because our study period coincided with the increase phase of a hare cycle (C. Krebs, University of British Columbia, pers. comm.), our results are consistent with suggestions that hare densities remain relatively low in southern British Columbia and the northwestern United States (Chitty 1950, Dolbeer and Clark 1975, Wolff 1980).

Food Habits

The large proportion (47%) of prey other than snowshoe hares documented during winter in this study to date is generally consistent with observations of opportunistic lynx food habits during hare lows (Koehler and Aubry 1994, Staples 1995, O'Donoghue et al. 1998*b*). The 35% combined proportion of red squirrel and flying squirrel kills also is roughly consistent with Koehler's (1990) reported 24% occurrence of tree squirrels in lynx scats collected year-round in Washington. Lynx predation on marten has not been previously reported and likely is opportunistic. Despite the substantial numeric quantity of alternate prey in the diet of lynx in this study, hares clearly represent the most important food source when biomass ratios are considered (e.g., 1:5 squirrel:hare; Nellis and Keith 1968), as is consistent with other lynx populations (Koehler and Aubry 1994, Staples 1995, O'Donoghue et al. 1998*b*). Although lynx in the Yukon cached partially consumed hares (O'Donoghue et al. 1998*b*), this behaviour was not observed in the southern Canadian Rockies.

Reproduction and Recruitment

Kitten production and survival were low during the study period. The apparent lack of recruitment to early winter among study animals is consistent with northern populations after a hare decline, when winter litter sizes were 0 (Brand et al. 1976, Poole 1994, Mowat et al. 1996). O'Donoghue et al. (1997) also did not detect family groups in their Yukon study area after the hare crash. Among the 4 uncollared family groups, litter sizes also were low in comparison to northern populations in the year preceding the hare decline, when mean winter litter sizes were 3.0 (n = 3; Poole 1994) and 3.3 (n = 12; Mowat et al. 1996). However, the occurrence of some kittens suggests that habitat quality varied among female home ranges within the study area. Koehler (1990) also reported low (12%) kitten survival to winter from 3 litters in north-central Washington. Evidence of breeding and den site abandonment of F/01 is consistent with the suggestion by Mowat et al. (1996) that breeding and implantation continues to occur regardless of prey densities.

Mortality

Survival of resident adults was high over both years, as was found in another population refugium (Slough and Mowat 1996). During hare lows in northern populations, most natural mortality is due to starvation and is preceded by nomadic movements (Ward and Krebs 1985, Poole 1994, Slough and Mowat 1996), as observed for 2 lynx in this study. These 2 deaths occurred during spring, in contrast to Poole's (1994) observation that most natural mortality in his Northwest Territories study area occurred during mid-winter. Evidence of cannibalism that we report has been documented before (Elsey 1954, Nellis et al. 1972, Brittell et al. 1989, Poole 1994, O'Donoghue et al. 1995, Slough and Mowat 1996).

Space Use and Movements

Annual home ranges of resident lynx in the southern Canadian Rockies were considerably larger than those reported for most other studies employing the same estimators, regardless of prey density or geographic locale (Koehler and Aubry 1994, Poole 1994, Slough and Mowat 1996). MCP home ranges in our study area were roughly 5–10 times larger than previously reported (Koehler and Aubry 1994, Poole 1994, Slough and Mowat 1996), although the influence of sample size on this estimator may confound comparisons (White and Garrot 1990). An exception to this was reported for the year after a hare crash in the Yukon, when 4 female and 2 male resident lynx maintained larger mean home ranges with high variability (Slough and Mowat 1996). Similarly, MDMs mostly were higher than in other studies of comparable methods (Ward and Krebs 1985, Poole 1994), implying greater foraging effort. One exception to this was documented by Ward and Krebs (1985) during a hare decline in the Yukon, when lynx increased their daily movements from 2.4 to 3.3 to 5.4 km/day as hares declined from 1.0 to 0.5 to 0.2/ha, respectively. However, another Yukon study did not find that lynx increased their active time in response to declining hare numbers (O'Donoghue et al. 1998*b*). Differences between 100% and 95% MCP home ranges reflect exploratory movements that were made throughout the year by all resident study animals. Physiography and the distribution of potential habitat may partially account for lynx home range size differences between geographic areas (Koehler and Aubry 1994). In comparison to other study areas, space use and movements by lynx in the southern Canadian Rocky Mountains therefore may be disproportionately greater

relative to prey densities.

Few studies have examined seasonal variation in space use by lynx. Although we found no significant seasonal differences in UD areas for either sex, tests were based on a low sample size. More extensive space use during summer was apparent for only 1 male; however, inclusion of the breeding period within the winter season may have masked seasonal differences among other lynx. Although not statistically tested, in Nova Scotia individuals of both sexes used areas during summer that were larger than those used during winter (Parker et al. 1983), and 2 females with young used smaller areas during summer in Alaska (Bailey et al. 1986). In accordance with the latter observation, the only lynx that made smaller daily movements during summer was a female with kittens.

Differential space use within home ranges may be influenced by habitat features and social interactions (Samuel et al. 1985). Obvious differences between annual UD levels indicate that lynx home ranges were not used homogeneously, but were associated with distinct core use areas. Poole (1994) also reported the use of obvious core home ranges by lynx (50% vs. 95% MCP). Evidence of intersexual spatial overlap that we report is consistent with most other populations (Koehler and Aubry 1994, Poole 1995). Sharing of home ranges between sexes or related females observed elsewhere also would be expected in the southern Canadian Rockies, considering the heterogeneous distribution of potential lynx habitat (compare Carr and MacDonald 1986, Sandell 1989).

Dispersal.— The timing of F/03's dispersal is within the March–June emigration peak observed by Slough and Mowat (1996) in the Yukon, but occurred 9 days before their earliest female kitten dispersal. The linear movements by this lynx and M/05 are in the lower extremes of those reported for lynx emigrating from northern populations (Slough and Mowat 1996, Poole 1997); however most of these were of adults. The orientation of movements relative to terrain and the Trans-Canada Highway suggest that dominant natural and human features may constrain dispersal options in the southern Canadian Rockies. The intensive use of 1 area exhibited by both lynx before their deaths does not imply residency, because dispersing felids have exhibited short-term use of temporary home ranges (Beier 1995, Apps 1996*a*).

CONSERVATION IMPLICATIONS

Although results are based on limited data, several conclusions specific to the southern Canadian Rocky Mountains can be made. Hare densities, and the diet, space use and movements, reproduction, subadult survival, and dispersal of lynx were consistent with those of northern populations during hare lows. Because the study period coincided with the increase phase of a snowshoe hare cycle, this suggests that lynx populations in the southern Canadian Rocky Mountains are not subject to the dramatic, cyclic pulses in productivity reported for northern populations. Further, our results suggest that the regional population presently exists at low densities, is of low productivity, and is patchy in distribution, as was also reflected by a regional trapper survey (Apps 1996*b*). Hence, we expect that lynx in the southern Canadian Rockies will be more vulnerable to mismanagement of habitat and populations than would otherwise be expected within northern ecosystems.

Conventional harvest management strategies allow for a more liberal take during the increase and high phase of the hare-lynx population cycle (Quinn and Parker 1987). However, where populations may not exhibit pronounced variation in productivity, conservative management should be adopted throughout the cycle. Moreover, the implications of localized overharvest may be severe where regional populations are patchy, and the spatial distribution of any harvest must be carefully managed. Localized populations are susceptible to extirpation, given the low densities and low recruitment with which they appear to be associated, and accelerated fragmentation at the regional level may inhibit the recolonization of suitable landscapes.

Productive and appropriately distributed population refugia are crucial to the regional conservation of wide-ranging carnivores, yet their existence for lynx in the southern Canadian Rockies is questionable. Across the regional landscape, human access levels and multiple resource demands (CORE 1994) on a limited land base with potential to support lynx suggest that de facto population refugia for this species do not presently exist in this region. The effectiveness of refugia requires land management that is consistent with lynx habitat, and it cannot be assumed that the management of "protected" areas has necessarily reflected such biodiversity values (e.g., Page et al. 1996). Resource development in our study area is focused within valley bottoms, or in broad, forested mountain passes, both of which have high potential to support resident lynx and to facilitate dispersal. Within such landscapes, forest management that promotes the attributes and distributions of stands preferred by lynx is required to sustain or enhance habitat effectiveness and connectivity, and it is important that options for such be maintained into the future. For example, to minimize risk from wildfire, facilities such as resorts, backcountry lodges, and campgrounds require permanent buffers of stands with little structural diversity and of limited value to communities with which lynx are associated. Hence, it is preferred that such developments be clustered rather than dispersed. In addition to forest management considerations, research into mitigation options is required where highways dissect lynx habitat, as they constrain lynx movements within our study area (Apps 1999).

Continued research will provide insight into lynx-habitat associations across spatial scales. Such information needs to be integrated with land-use planning at ecologically relevant

scales, requiring coordination among agencies and management jurisdictions. Although we do not advocate single-species management, lynx tend to be associated with forest mosaics of stand and landscape structure that may resemble conditions under a natural disturbance regime. Therefore, they may represent an ideal "focal-species" for assessing the efficacy of biodiversity management strategies, at multiple spatial and temporal scales, within montane and subalpine ecosystems of the southern Canadian Rockies.

ACKNOWLEDGEMENTS

These first 2 years of this ongoing study were funded by the British Columbia Habitat Conservation Trust Fund, Parks Canada, the Columbia Basin Fish and Wildlife Compensation Program, and the Habitat and Silviculture Assistance Fund. Field assistance was capably provided by R. Owchar, B. Bertch, S. Stevens, P. Wandeler, S. Hawes, D. Quinn, B. Baxter, C. Darimont, M. Vassal, J. Weaver, D. Poszig, and D. Ghikas. J. Weaver and G. Mowat also contributed helpful advice on study design and methods. Partial administration of this study has been provided by J. Krebs of the Columbia Basin Fish and Wildlife Compensation Program. Veterinary services were provided by T. Shury and M. Zehnder. Temporary animal holding and care were facilitated by T. Kinley and N. Newhouse.

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The Endangered Species List and Lynx Conservation in Idaho: Public Paranoia and Politics Over Biology

D.L. Murray
Department of Fish and Wildlife Resources
University of Idaho, Moscow, ID 83844 USA
dmurray@novell.uidaho.edu

ABSTRACT

The University of Idaho, in cooperation with the Idaho Department of Fish and Game and the United States Forest Service, had planned to release lynx to the Clearwater National Forest (CNF) in Idaho. Lynx are at the southern periphery of their range in Idaho, and although once present on the CNF, now are apparently extinct or very rare in the area. The United States Fish and Wildlife Service is currently reviewing the status of lynx in the contiguous United States and may decide to add the species to the Endangered Species List (ESL). If the species does become listed, measures to protect native lynx may be implemented, including possible road closures and restrictions on activities in natural areas. Preliminary work undertaken on the CNF in anticipation of the release suggested that the area likely could support a viable population of lynx. Also, the Fish and Wildlife Service ruled that released lynx would not be covered under the Endangered Species Act because of their non-native status. However, opposition to the project was mounted by select conservation groups and individual hunters; as a result of such pressure the State of Idaho eventually refused to issue an import permit for lynx. Thus, lynx will not be released in Idaho in the foreseeable future. In this presentation I will describe the chronology of events that lead to the permit refusal, and show how in this instance the threat of listing lynx under the Endangered Species Act acted as a catalyst for public opposition. I will conclude by providing evidence to illustrate how such opposition was based on public ignorance and/or intransigence, and highlight the need for aggressive lobbying in future instances of conservation of potentially controversial species.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb ,1999. Volume Two
B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Bird in Hot Water: Responses by Marbled Murrelets to Variable Ocean Temperatures off Southwestern Vancouver Island

Alan E. Burger
Department of Biology, University of Victoria
Victoria, BC, V8W 3N5, Canada
aburger@uvvm.uvic.ca

ABSTRACT

Periodic warm-ocean episodes are known to have negative impacts on some seabirds in the Pacific Northwest. Effects on the threatened marbled murrelet (*Brachyramphus marmoratus*) are not known. I review the responses of the murrelet to variable nearshore temperatures for 1979 through 1998, which includes 3 El Niño–Southern Oscillation (ENSO) events and a general warming trend. Counts of murrelets in systematic marine transects at 5 sites in British Columbia covering 4–8 years per site showed considerable intra- and inter-seasonal variation, with negative effects of warm seas evident at 3 sites. Samples were generally too small for rigorous statistical testing. Inland surveys over 8 years in old-growth forests in Carmanah-Walbran showed significant effects of ocean temperature. Specifically, the mean frequency of occupied detections, a measure of near-nest flight behaviours, was strongly negatively correlated with nearshore ocean temperature. Surveys in the Ursus Valley, Clayoquot Sound, over 4 years showed a similar, but not significant, trend. Although the link between occupied detections and nesting success is unclear, these results suggest that warm nearshore conditions somewhat inhibit breeding activity. Reduced prey availability in warm seas is a likely cause. This finding has important implications for conservation and monitoring of this species. Inland surveys of flight detections are often used to delineate and map suitable habitat for murrelets as part of logging plans, and could be seriously flawed if they sampled years with depressed activity. If murrelet breeding is inhibited by warm oceans then global warming, coupled with continued loss of nesting habitat in old-growth forests, is likely to lead to significant population declines. Most research, monitoring, and management focus on their forest nesting habitat, but marbled murrelets are seabirds, and changes in their marine environment require equal consideration.

Key words: at-sea surveys, *Brachyramphus marmoratus*, El Niño, inland surveys, marbled murrelet, ocean temperature effects, Vancouver Island.

The marbled murrelet (*Brachyramphus marmoratus*) is a widespread seabird, which nests solitarily in large old-growth trees through most of its range (Hamer and Nelson 1995, Nelson 1997). It is listed as threatened in Canada and the United States and is on the British Columbia Red List (i.e., a species likely to become endangered if threats are not dealt with and trends reversed). Loss of nesting habitat from logging is the primary threat through its range, but oil spills, gill nets, and increased predation at nests are additional threats (Ralph et al. 1995a). In this paper I analyze the effects of variable nearshore sea temperatures on counts of murrelets in marine census transects, and on detections of murrelets in inland forests. I use data from several multiyear studies on southwestern Vancouver Island, which supports one of the highest densities of murrelets south of Alaska (Burger 1995a,b, Burger et al. 1997, Rodway and Regehr 1998, Bahn et al. 1998).

Changing ocean temperatures profoundly influence marine ecosystems in the northeast Pacific (McGowan et al. 1998), including British Columbia (Freeland 1992). Variable sea temperatures and El Niño events are known to affect seabirds in the northeast Pacific (e.g., Vermeer et al. 1979, Hodder and Graybill 1985, Hatch 1987, Ainley and Boekelheide 1990, Wilson 1991). The effects on marbled murrelets are not well known, but the density and distribution of the murrelet's prey are likely to be affected by ocean temperatures (Ainley et al. 1995, Ralph et al. 1995a).

The solitary, tree-nesting behaviour of marbled murrelets precludes censusing and monitoring of breeding success at the nest site, as is usual for most other seabirds. Consequently, specialized research and monitoring techniques have been developed for both marine and inland

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

habitats (Ralph et al. 1994, RIC 1997). Boat transects in nearshore seas are used to estimate population densities, marine habitat preferences, and juvenile recruitment (e.g., Sealy and Carter 1984, Carter and Sealy 1990, Becker et al. 1997, Kuletz and Kendall 1998). Inland, the presence and relative densities of marbled murrelets are determined by recording visual and auditory detections made during standardized surveys at dawn, when the birds are most active in and above the forests (Rodway et al. 1993*a*,*b*; Ralph et al. 1994, 1995*a*). The inland surveys are routinely used to assess occupancy and the relative importance of forest stands, and hence have important implications for land management and timber harvests in the Pacific Northwest. Annual variations in inland detections and possible effects of marine influences have seldom been studied (Ralph et al. 1995*a*,*b*). There are compelling reasons to understand the responses of marbled murrelets to ocean conditions, because data indicate long-term warming in coastal waters, including British Columbia (Thomson et al. 1984, Freeland 1992, McGowan et al. 1998).

METHODS

Sea Temperatures

Monthly mean sea surface temperatures from Amphitrite Point light station were obtained from the Institute of Ocean Sciences, Sidney, B.C. (http://www.ios.bc.ca/ios/osap/data/). This light station provided a 63-year series of daily temperature records, which are useful indicators of nearshore ocean conditions off southwest Vancouver Island (Freeland 1992). I used mean temperatures for the period April through July, which is the time of greatest marine production in this region, and includes the core of the murrelet's breeding season. Temperature anomalies (positive for unusually warm water) were calculated as the difference between the mean monthly temperatures during the study period and the monthly means from the complete 63-year series.

Marine Surveys

In and near Barkley Sound, marbled murrelets were counted along 4 repeatedly sampled transects: Trevor Channel (43 km), the Broken Group Islands (8.8 km), the West Coast Trail (66.6 km), and the Cape Beale survey route (19.5 km; Fig. 1). Methods differed somewhat among the areas, but were consistent within each area. Small boats (4-6 m long; observer eye height 1.5–2.5 m above the sea) were used for the Trevor Channel and Broken Group transects, and larger vessels (8–12 m long; eye height 2.0–3.5 m) for the West Coast Trail and Cape Beale routes. In all transects the observers scanned ahead and on both sides of the vessel, while travelling at 15–20 km/hour, slowing to count larger groups of birds or identify plumage types. Two or more

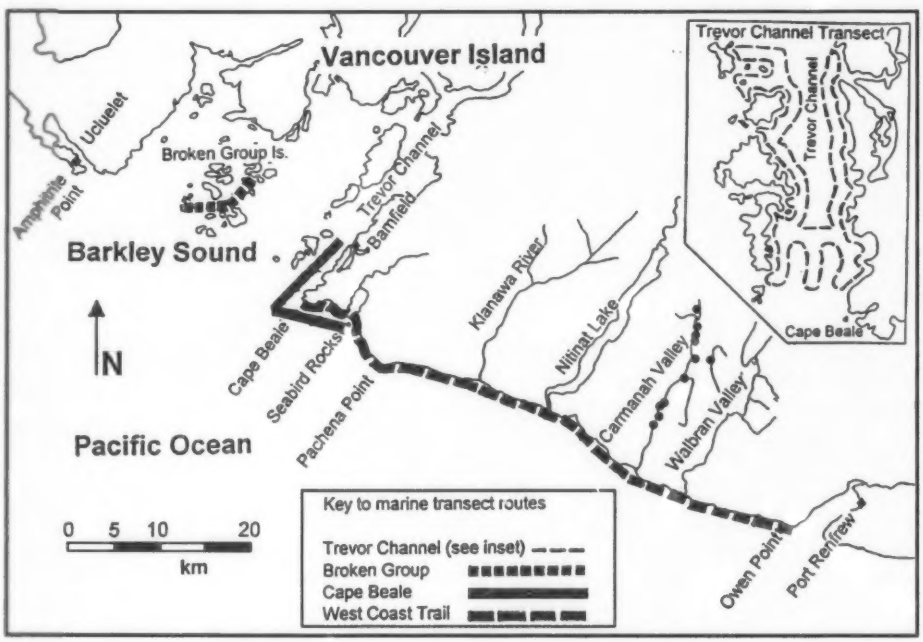

Figure 1. Study area on southwest Vancouver Island, showing the locations of the Amphitrite Point light station, 4 marine transect routes, and the 11 inland stations in the Carmanah–Walbran watersheds (solid dots).

observers were generally used, but in a few Trevor Channel and Broken Group surveys a single observer recorded sightings on a tape recorder. Binoculars were used to confirm identifications and plumage types, but were not used to scan for murrelets. All surveys were undertaken when the sea was relatively calm (Beaufort scale 3) and it was not raining heavily. Surveys were completed in the mornings, when counts in this region were least variable (Carter 1984). In most surveys, birds within or beyond strip boundaries 150 or 200 m on either side of the vessel were recorded separately; however, for consistency, I used all birds visible on either side of the vessel. The West Coast Trail data included birds seen on the water and flying, but all other data were birds on the water only. Observers were trained in identification of birds and census methods, and most were experienced seabird biologists.

The Trevor Channel survey covered the same area sampled intensively by Carter (1984), including sheltered nearshore water and an exposed channel. The Broken Group route covered sheltered water among islands. The West Coast Trail transect ran 200 m offshore in shallow nearshore sea exposed to the open Pacific Ocean. The Cape Beale route, also sampled by Carter (1984), covered open channel and exposed, deeper waters.

I also compared data from boat surveys made in Clayoquot Sound in 1982 by Sealy and Carter (1984) and repeated in the 1990s by Kelson et al. (unpublished data, 1995) using the same methods.

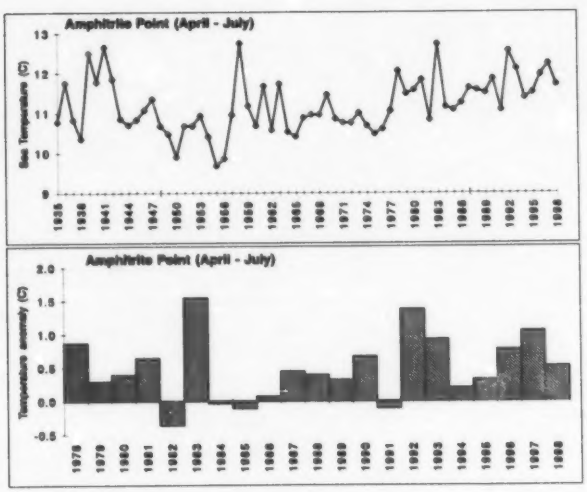

Figure 2. Sea surface temperatures recorded at Amphitrite Point. The upper graph shows the mean temperatures during April through June in each year, and the lower graph shows the temperature anomaly (deviation from the long-term average) from 1978 to 1998.

Inland Audiovisual Surveys

Data were analyzed from 11 inland stations sampled in each year in the Carmanah–Walbran watersheds (Fig. 1; Burger 1994) and 12 stations sampled annually in the Ursus Valley in Clayoquot Sound (Bahn and Newsom 1999). Each station was sampled 2–4 times per year, at >10 day intervals, and annual means were calculated for each station. All stations were in continuous, unlogged, old-growth forest in the Coastal Western Hemlock biogeoclimatic zone and the West Vancouver Island windward maritime and montane ecoregions. Vegetation of these stands was analyzed in Burger (1994), Bahn (1998), and Rodway and Regehr (1998). Trained observers recorded murrelet activities during 2-hour (minimum) dawn watches, using the standardized fixed-station protocol (Ralph et al. 1994, Resources Inventory Committee 1997). No surveys were conducted if it was raining heavily.

The unit to measure inland murrelet activity was the "detection," defined as "the sighting or hearing of one or more murrelets acting in a similar manner" (Paton 1995). A subset of these, named "occupied detections," represented behaviours likely to occur in near nests, including: birds flying into, out of, through, or below the forest canopy; birds perching, landing, or attempting to land in trees; birds calling from a stationary position in the canopy; and birds circling above or below the canopy (Ralph et al. 1994, Paton 1995).

Radar Counts

Murrelets entering the Bedwell and Ursus watersheds were counted at dawn using a high-frequency marine surveillance radar stationed at the mouth of the Bedwell River. This method, described by Burger (1997), estimates of the total number of murrelets, breeding and nonbreeding, entering the watersheds. Surveys in which there was rain for >10 minutes during the peak of pre-sunrise activity were discarded. About 75% of the murrelets counted at the Bedwell mouth were estimated to be destined for the Ursus Valley.

RESULTS

Sea Surface Temperatures

Mean sea temperatures at Amphitrite Point for the spring and early summer (April through July) showed considerable fluctuations over the years, but there has been an increasing trend since the late 1970s (Fig. 2). From 1978 to 1998, mean temperatures were above average, with the exception of 4 years (1982, 1984, 1985, and 1991). Peaks of high temperature associated with El Niño events were evident in 1983, 1992–1993, and 1997.

Marine Surveys

Survey data from the core of the breeding season, 15 May through 16 July were selected. Counts of marbled murrelets in and near Barkley Sound in 4 survey areas varied considerably among years (Fig. 3). No significant correlations were found between murrelet numbers and sea surface temperature in any of the 4 survey areas. In all 4 survey areas, counts were consistently low in 1993, which was the second successive year with high sea temperatures.

The mean counts made by Carter (1984) in 1979 on the Cape Beale transect and in 1980 in Trevor Channel were higher than those made between 1987 and 1998 in these areas (Fig. 3). This result might be due to an overall decline in murrelet densities over the intervening years, or to changes in ocean conditions. Carter's counts were done when sea temperatures were similar to those experienced in

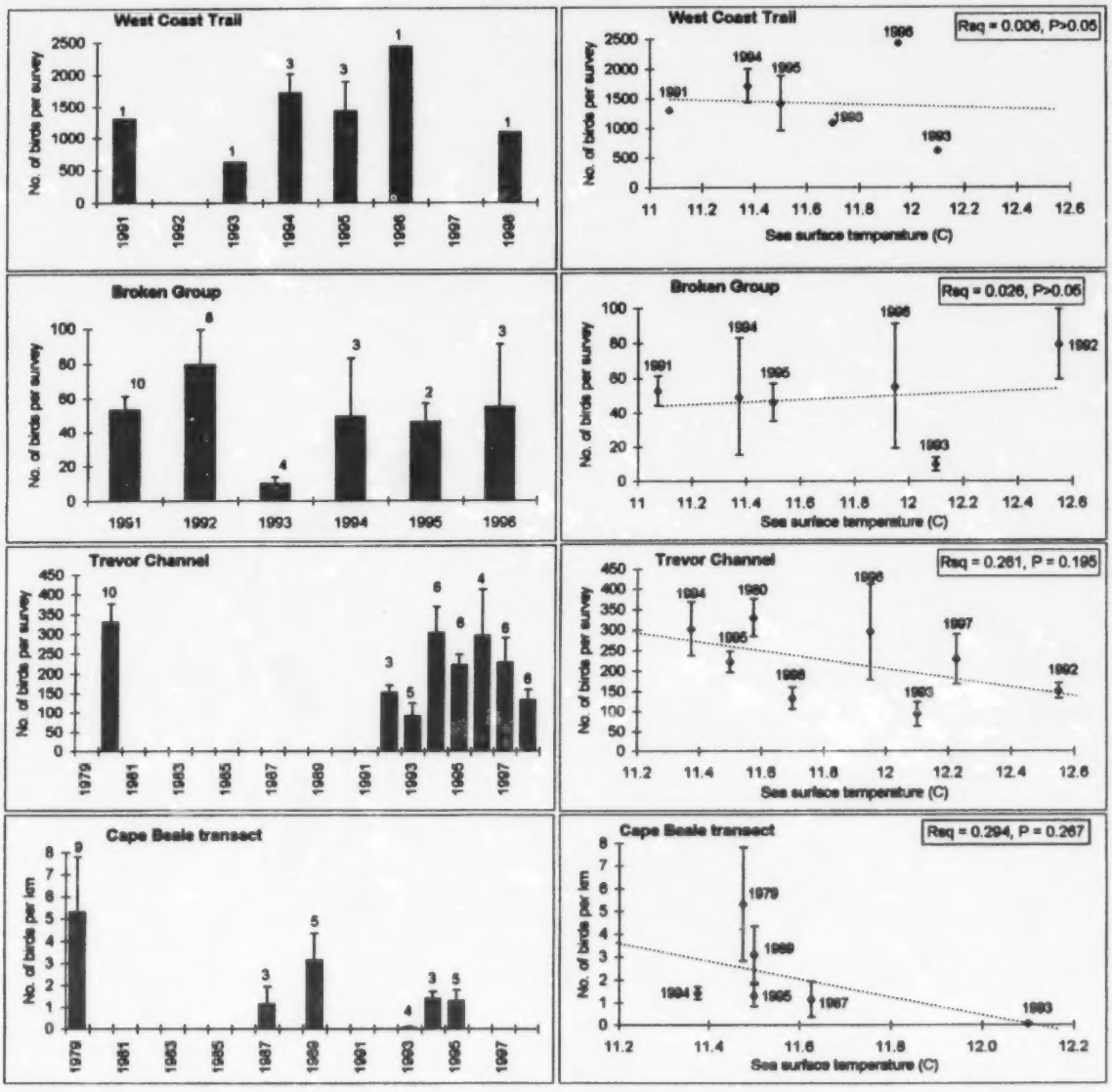
West Coast Trail
No. of birds per survey
Sea surface temperature (C)
Rsq = 0.006, P>0.05
Broken Group
Rsq = 0.026, P>0.05
Trevor Channel
Rsq = 0.261, P = 0.195
Cape Beale transect
No. of birds per km
Rsq = 0.294, P = 0.267

Figure 3. Counts of marbled murrelets in marine transects in and near Barkley Sound. The graphs on the left show annual means (±SE) from 15 May through 16 July. The sample sizes are the numbers of transects; years with no data had no transects. The right hand graphs show the same data plotted against mean sea temperatures. None of the correlations were significant and the dotted lines are the trend lines and not significant regressions.

1987–1998 (Fig. 3), suggesting that temperature effects, if any, were not entirely responsible for the changes.

Counts of murrelets made in Clayoquot Sound declined between 1982 and the 1990s, which was attributed largely to the effects of logging in the area (Kelson et al. 1995). Closer inspection of the data revealed that changes were evident only in the exposed inshore areas, and not in the sheltered channels (Fig. 4). Sea temperatures in 1982 were unusually cool (Fig. 2), and there was a significant negative correlation between counts of murrelets and sea temperature, which was entirely due to the changes in the exposed inshore areas (Fig. 4). It was not possible with this small sample to separate the effects of time (changes over the years) and sea temperature, but both clearly need to be considered if larger samples become available.

Inland Audiovisual Surveys

Mean values of total detections in Carmanah–Walbran were relatively constant through the 8 years sampled, with the exception of high counts in 1994 (Fig. 5). Total detections were not significantly correlated with sea surface temperatures. Occupied detections, a better estimate of near-nest activities, varied more widely, with particularly low frequencies in 1992, 1993, and 1998. Occupied detection frequency showed a significant, negative correlation with mean sea temperatures, and 64% of the variation in occupied detection frequency could be explained by variations in sea temperature

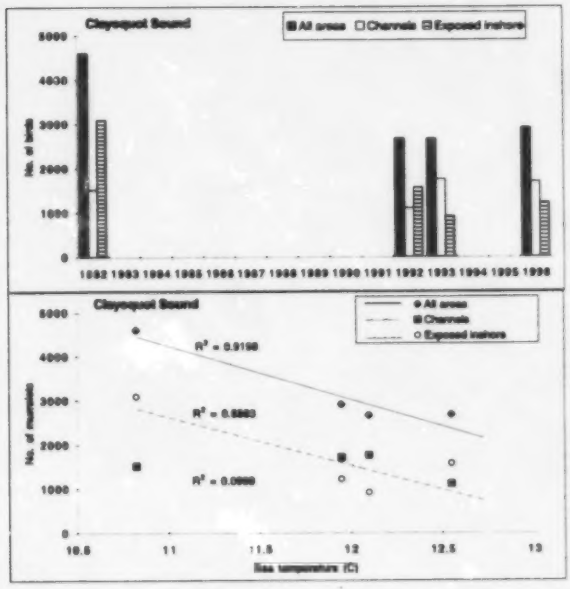

Figure 4. Counts of marbled murrelets in Clayoquot Sound (data from Sealy and Carter 1984; Kelson et al. 1995; J. Kelson, unpubl. data). Data from sheltered channels in the sound were plotted separately from those from the more exposed, inshore areas.

(r^2 = 0.64) (Fig. 5).

The sample of inland surveys in the Ursus Valley (4 years) was not adequate for rigorous statistical testing (Fig. 6). As in Carmanah–Walbran, the occupied detections showed greater variation than total detections. Both measures showed some decline with increasing sea temperature, but these correlations were not statistically significant.

Radar Counts

The numbers of marbled murrelets entering the Bedwell–Ursus watersheds varied among the 4 years sampled (Fig. 6). The mean counts declined with increasing sea temperatures, but this trend was not statistically significant.

DISCUSSION

Responses by Marbled Murrelets to Variable Ocean Conditions

Fluctuations in the physical environment of the northeast Pacific occur at several time and spatial scales. Significant inter-decadal changes in temperatures, currents, and climate have been identified, accompanied by large biological changes, termed regime shifts (McGowan et al. 1998). These include changes in densities, distribution, recruitment, and food webs, affecting a wide range of organisms. Imposed on these decadal changes are shorter events, lasting months or a few years, including ENSO events. This study was done during a 20-year warm phase, in which 3 shorter El Niño events occurred. A detailed review of ocean processes off Vancouver Island is beyond the scope of this paper, but probably no "normal," stable condition exists in marine ecosystems. The biology of marbled murrelets, like other marine organisms, changes frequently at different time and spatial scales.

In general, warmer nearshore temperatures off southwestern Vancouver Island seemed to have negative effects on marbled murrelets. Counts at sea showed a negative trend with temperature at 3 of the 5 survey locations. This finding was statistically significant at only 1 location, but all the sample sizes were small and intra-seasonal variations were large, reducing the chances of detecting significant trends. Inland, murrelet detections showed a negative trend with temperature in both the Carmanah–Walbran and Ursus samples. This negative correlation was significant for occupied detections in Carmanah–Walbran, the location with the longest data series. Radar counts at the Bedwell showed a similar trend, but the sample was too small for rigorous statistical testing.

The present data suggest that during unusually warm years murrelets moved out of the sampled marine areas and performed fewer near-nest occupied behaviours in the forests. Longer series of marine and inland surveys are needed to adequately test these patterns. The data were all

obtained in the 1980s and 1990s, during a prolonged warm phase in the northeast Pacific, and so do not sample the full range of ocean conditions experienced off Vancouver Island. Furthermore, populations of marbled murrelets in this area have been affected by the loss of nesting habitat caused by intensive logging of old-growth forests over the past 2 decades (Sealy and Carter 1984, Rodway et al. 1992, Burger 1995*b*, Kelson et al. 1995). Separating the effects of ocean temperatures and declines due to logging is problematic, particularly for the Clayoquot Sound marine counts (Fig. 4).

Possible Causal Factors

In the northeast Pacific, warm water (El Niño) events are associated with reduced upwelling, lowered productivity of phytoplankton and zooplankton, and reduced recruitment and availability of small schooling fish (Wooster and Fluharty 1985, Freeland 1992, McGowan et al. 1998). These changes often have negative effects on seabirds, including dietary changes, reduced local densities of birds at sea and in colonies, lower breeding success, failure to breed, and increased mortality (Vermeer et al. 1979, Hodder and Graybill 1985, Bayer 1986, Ainley and Boekelheide 1990, Wilson 1991). The effects of El Niño conditions on marbled murrelets have not been described, apart from 1 study off central California which included few murrelets (Ainley et al. 1995). In this study, murrelets were least abundant during El Niño periods, and in all years their spatial distribution was negatively correlated with sea temperature.

During the breeding season, marbled murrelets off southwest Vancouver Island prey mainly on small schooling fish, predominantly sand lance (*Ammodytes hexapterus*) and juvenile herring (*Clupea harengus*) (Carter 1984; A. E. Burger, pers. observation). Other small schooling fish (e.g., juvenile salmonids), and euphausiids (mainly *Thysanoessa spinifera*) were also likely to be important prey (Burkett 1995).

Several processes might have affected prey availability off Vancouver Island during unusually warm years. First, densities and recruitment of *Thysanoessa spinifera* were sensitive to temperature, and declined significantly in and near Barkley Sound during the 1992 El Niño event (Tanasichuk 1998a,b). This decline could have affected murrelets directly if they fed on euphausiids, or indirectly because euphausiids were important prey for schooling fish. Second, large schools of mackerel (*Scomber japonicus*) and jack mackerel

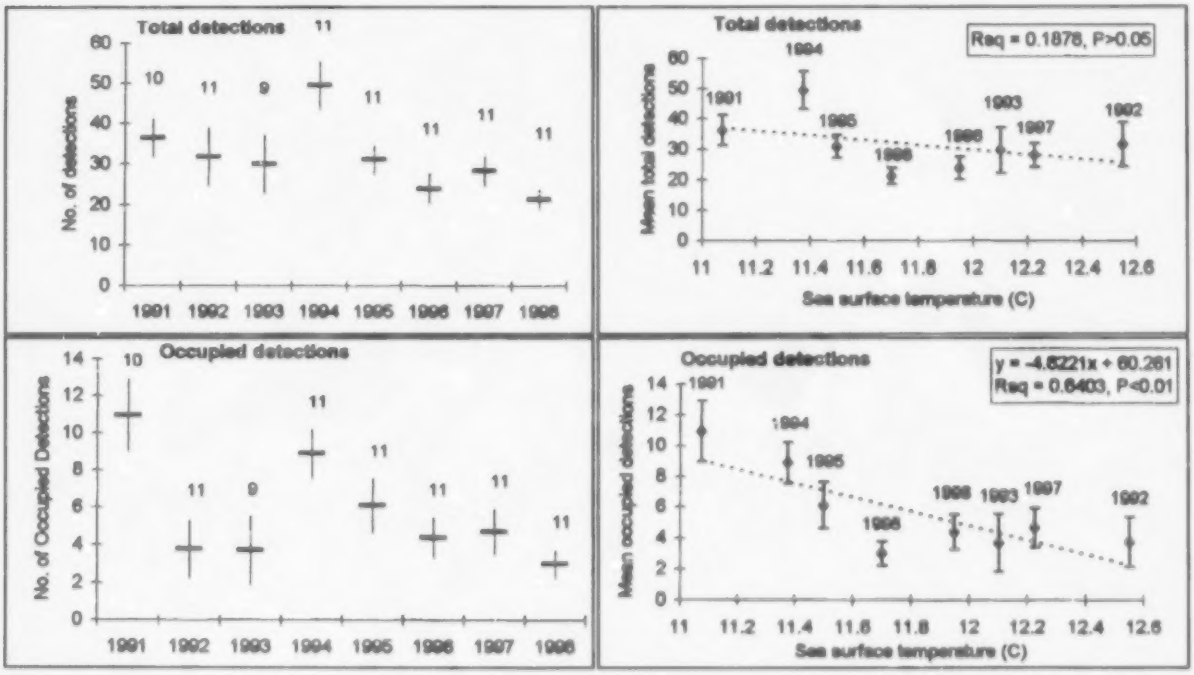

Figure 5. Mean frequencies (±SE) of all detections (total) and occupied detections recorded at 11 stations in the Carmanah–Walbran watersheds in each year, during the period 15 May through 16 July. The left graphs show the annual means and the right graphs show the same data plotted against mean sea temperatures. Occupied detections were significantly correlated with temperature and the regression line and equation are shown. Total detections were not significantly correlated with temperature. The dotted line indicates the trend, not a significant regression.

(*Trachurus symmetricus*) invaded Barkley Sound and adjacent seas during the warm years in the 1990s. These species eat similar prey to that taken by marbled murrelets (Hart 1973), but are usually uncommon off Vancouver Island (Hay et al. 1992). The mackerel decimated local stocks of juvenile salmonids and juvenile herring (B. Hargreaves, pers. comm.). Third, unusually warm water might also reduce the availability of existing prey to murrelets. In particular, sand lance avoid unusually warm surface layers (Field 1988); this might make them inaccessible to murrelets in deep water (>40 m) (Mathews and Burger 1998). Finally, the sightings of schools of Pacific sardines (*Sardinops sagax*) increased noticeably off southwest Vancouver Island through the late 1990s. Sardines in British Columbia are usually adults (Hart 1973) and probably too large for murrelets to eat. Their presence might, however, cause declines in other smaller schooling fish which the murrelets eat.

Conservation and Management

Consequences of Variable Ocean Conditions

If marbled murrelets are sensitive to changes associated with warmer sea temperatures, there are important consequences for conservation and management. In the short term, the effects of sea temperature on detections made during audio-

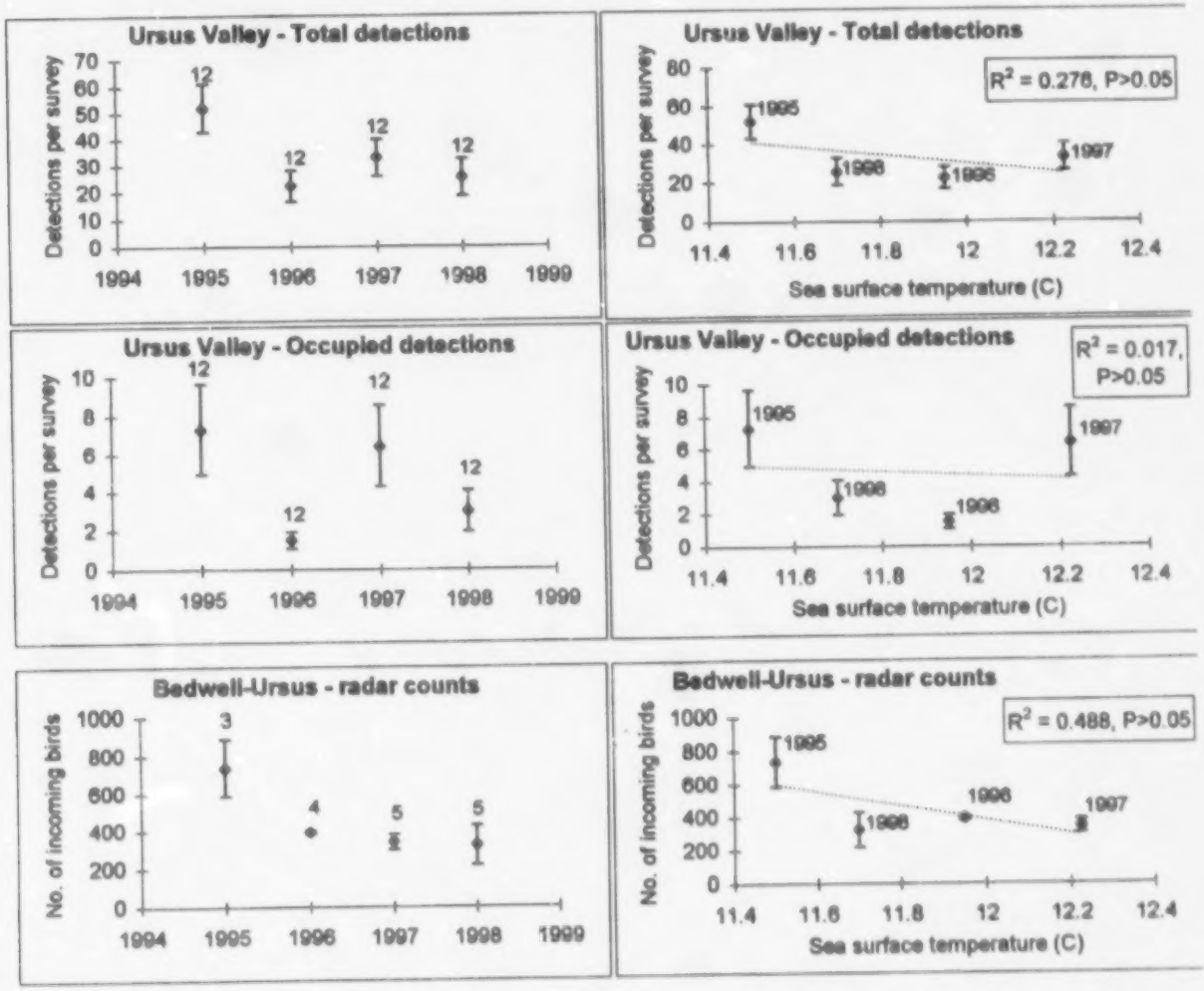

Figure 6. The left graphs show the annual mean frequencies (±SE) of all detections (total) and occupied detections recorded at 11 stations in the Ursus Valley, Clayoquot Sound, and the count made with radar of murrelets entering the Bedwell–Ursus watersheds. The graphs on the right show the same data plotted against sea temperatures. None of the correlations were significant and the dotted lines show the trends only.

visual surveys must be carefully considered. Audiovisual surveys are the standard method for assessing the presence and relative densities of murrelets, and occupancy of stands in the forests (Ralph et al. 1994, 1995*a*, Resources Inventory Committee 1997). Many management decisions are based on 1 or 2 seasons of audiovisual surveys. If these surveys coincided with warm years with few occupied detections, the distribution, occupancy, and relative densities of murrelets could be seriously underestimated (Burger 1995*b*). This type of error could be avoided by completing multiyear surveys, or at least by comparing detections with local sea temperatures and other evidence of prey availability.

Global warming has long-term implications on marbled murrelets in British Columbia. For example, the regression in Figure. 5 indicates that there would be no occupied detections if the nearshore sea temperature exceeded 13°C. This interpretation is probably simplistic, but such temperatures are possible and could have serious negative effects on murrelets. Unsuitable marine conditions could cause emigration or reduced breeding activity even if suitable nesting habitat remained in the forests. The combination of unsuitable foraging conditions at sea and continued loss of nesting habitat inland could severely reduce murrelet populations. Most conservation and management of marbled murrelets are focused on their forest nesting habitat, but we should not forget that these are seabirds, dependent on productive nearshore marine ecosystems. Conservation strategies for this species need to integrate both the marine and inland ecological requirements.

ACKNOWLEDGEMENTS

The data used here were gathered by many people working for the University of Victoria and the British Columbia Ministry of Environment, Lands and Parks. In particular, I thank J. Austin, V. Bahn, C. Darimont, S. Dechesne, A. Eriksson, J. Kelson, A. Lawrence, I. Manley, D. Newsom, A. Stewart, R. Shoop, A. Tillmanns, the wardens of Pacific Rim National Park, students and staff from Bamfield Marine Station, and volunteers and staff of the Western Canada Wilderness Committee. T. Chatwin, D. Doyle, L. Jones, and B. Redhead helped get projects funded and running. I gratefully acknowledge the funding received from: Forest Renewal British Columbia; the Endangered Species Recovery Fund (World Wildlife Fund Canada and Canadian Wildlife Service); Habitat Conservation Trust Fund (British Columbia Ministry of Environment, Lands and Parks); Heritage Canada (Pacific Rim National Park); Canadian Wildlife Foundation; and Friends of Ecological Reserves.

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Effects of Forest Edge and Canopy Cover on the Detection and Nesting Ecology of Marbled Murrelets

I.A. Manley
Department of Biological Sciences, Simon Fraser University
8888 University Dr., Burnaby, BC, V5A 1S6, Canada
iamanley@sfu.ca

I. Blackburn
Ministry of Environment Lands and Parks
Lower Mainland Region
10470 152nd St., Surrey, BC, V3R 0Y3, Canada

ABSTRACT

In order to define the nesting habitat of marbled murrelets, it is important to understand how edges and openings affect their nesting ecology. We investigated the effect of clear-cut edges and canopy openings on the activity, nesting density and reproductive success of marbled murrelets on the Sunshine Coast. Using RIC inventory methods (audio-visual detections), we found higher murrelet detections at forest edges than at paired locations 250 m from edges. However, using tree-climbing search method we found nesting density was two times higher at forest interior sites than at edges. This suggests that audio-visual surveys are biased by increased visibility of birds in open areas. We examined how habitat scale influences selectivity for cover and openings at nest sites. Murrelets selected nest sites on large, mossy branches with high vertical cover. Nest trees were adjacent to significantly larger canopy openings required to access the nest site. However, murrelets selected nest patches and stands with high canopy closure. This suggests that selection for openings does not outweigh their need for cover. Cover is important because murrelets have low nesting success (20-28%) largely due to predation. Successful nests were located significantly farther from forest edges (254 m vs. 79 m). We conclude that although surveys may detect more birds at forest edges, the forest interior provides critical habitat with a higher nesting density and reproductive success for marbled murrelets.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999.

Marbled Murrelet as Target Species For Land Management in Coastal British Columbia

Volker Bahn
Central Westcoast Forest Society
P.O. Box 364, Ucluelet, BC, V0R 3A0. Canada
vbahn@hotmail.com

Deanna N. Newsom
Central Westcoast Forest Society
P.O. Box 364, Ucluelet, BC, V0R 3A0, Canada

ABSTRACT

Coastal temperate rain forests are complex ecosystems with high economic value. Conservation management of these forests must be based on clear, defendable strategies with quantifiable goals if they are to withstand economic pressure for exploitation. We introduce the target species strategy as an efficient management tool which allows quantification of conservation goals and continuity in planning time frames. A target species is a species used in defining and monitoring conservation goals. The marbled murrelet (*Brachyramphus marmoratus)* is an excellent example of a target species. This seabird is highly dependent on coastal old-growth forests as breeding habitat. It is a threatened species in Canada and is considered for legal designation as threatened or endangered in British Columbia. Therefore, it has become a focal species in coastal temperate rain forest conservation. Based on data collected during 4 years of marbled murrelet inventory by the British Columbia Ministry of Environment, Lands and Parks in Clayoquot Sound, British Columbia, we designed a habitat suitability index for the marbled murrelet for efficient habitat evaluation. This index allows a prioritization of habitats based on information from digital Vegetation Resources Inventory maps recently completed in Clayoquot Sound, as well as fine scale habitat prioritization based on vegetation plots. The habitat suitability index was used with a geographic information system (GIS) to rank and map habitats of importance to marbled murrelets in the Clayoquot Sound. The target species strategy, in combination with the presented habitat evaluation tools, bridges the gap between research and conservation management of the marbled murrelet and its habitat.

Key words: *Brachyramphus marmoratus*, Clayoquot Sound, coastal temperate rain forest, habitat suitability index, marbled murrelet, target species.

The primary coastal temperate rain forest is a unique and rare ecosystem. It originally covered only <0.2% of the world's land surface; today, it is threatened by permanent deforestation and conversion to managed forests. More than 50% of this rain forest worldwide has already been degraded (Bryant et al. 1997). This ecosystem is highly productive and contributes largely to global biodiversity (Scientific Panel for Sustainable Forest Practices in Clayoquot Sound 1995, Bunnell and Chan-McLeod 1997).

One-quarter of the world's remaining old-growth temperate rain forest is in British Columbia (Bryant et al. 1997). Some 53% of British Columbia's and 70% of Vancouver Island's old-growth temperate rain forest has been logged already (Sierra Club of British Columbia 1997), and logging or development has degraded more than two-thirds of it in British Columbia (Bryant et al. 1997). Most of the remaining rain forest, except for protected areas, is slated to be logged in the next 10 years (Bryant et al. 1997).

Clayoquot Sound has the largest contiguous stretch of temperate rain forest on Vancouver Island. It encompasses the southern-most multi-watershed complex of pristine rain forest in North America. To improve land management, the Scientific Panel for Sustainable Forest Practices in Clayoquot Sound (1995) produced many recommendations on planning processes, forest practices, and monitoring activities. Clayoquot Sound is supposed to become an economic, social, and ecological model of sustainable forestry.

The focus of the monitoring recommendations of the Scientific Panel was habitat and structural characteristics. An important goal was ecosystem integrity. Concepts such as ecosystem integrity or health have become increasingly widespread in recent years. If they are used in a management context (i.e., for land on which human use and conservation coincide), these concepts need to be clearly defined, quantified, and monitored to be meaningful and operable.

Although we agree that monitoring habitat is important for landscape-level planning, monitoring the organisms that live in the habitat is likely the best indicator of its state (Simberloff 1998). Furthermore, species are entities that do

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Table 1. Selection criteria for target species (from Bahn 1998).

Required characteristics	Preferred characteristics
indigenous to the region	important to First Nations
has a chance to survive through the next planning periods without artificial support	threatened (mostly by habitat depletion not directly by human-induced mortality)
representative for the needs other species (umbrella effect)	complex and high habitat requirements (specialist, not ubiquitous)
sensitive to habitat alterations	geographically restricted distribution
	indicator, keystone, flagship species
	not migratory
	low dispersal
	ecological and demographic information available
	feasibility of research
	economically significant

not change in planning time frames and that can be counted. Hence, we would like to put forward a species-based strategy to define and quantify conservation efforts. This strategy could use the existing research results on several selected groups of species in the area (e.g., marbled murrelets, Roosevelt elk, black bears, bats, amphibians, aquatic invertebrates, and owls).

This approach is not meant to replace landscape-level planning which clearly must rely on protection of a representative set of ecosystems without any need to count or prove anything. The species-based approach is only meant for management in areas where human use and conservation coincide (which should be all used areas outside of protected areas).

THE TARGET SPECIES STRATEGY

Most land management and conservation strategies that we know of have a striking similarity: the arguments that their critics use to disqualify the approach. Common criticisms include

- lack of precise definitions of strategy goals (e.g., ecosystem health or integrity);
- lack of scientific evidence to support the underlying ideas (e.g., umbrella or keystone species whose protection is assumed to protect other species and ecological processes);
- too complex to be handled with available resources;
- oversimplifying the complicated nature of ecosystems; and
- subjectivity in its interpretation and dependence on current human values.

The universal applicability of these criticisms makes one wonder whether the problems might be inherent in any attempt to manage ecosystems with the relatively small resources typically available to conservation endeavours, rather than having anything to do with the strategies themselves. We admit that all but the last criticism apply to the strategy we will promote as well. Therefore, we want to focus on the benefits of the target species strategy rather than the known drawbacks.

Several related strategies aim to manage land by managing a selection of species: target species (Hansen et al. 1993, Walter et al. 1998); umbrella species (Launer and Murphy 1994); array of indicator species (Plachter 1991*b*); keystone species (Paine 1966); management indicator species (Woodruff 1989); focal species (Lambeck 1997); flagship species (Yen 1993); and indicator species (Soulé and Kohm 1989). Essentially, all of these strategies aim to prevent habitat degradation and further loss of biodiversity by monitoring selected species and maintaining a minimal population level. The strategies differ in how these species are selected, what they stand for, and at what levels they are protected or monitored. We will use the term target species (Woodruff 1989, Walter et al. 1998) and will use the following definition: *A target species is a species used in defining and monitoring conservation goals.*

Management strategies, such as ecosystem sustainability, health, or integrity, will remain weak and hard to implement without quantification of goals and environmental quality standards (Plachter 1991*b*, Hansen et al. 1993). By setting certain lower limits of tolerance for population size of the target species, environmental quality goals become clearly defined and measurable (Plachter 1996, Heidt et al. 1997, Walter et al. 1998).

The selection process for target species should follow certain rules which would have to be adapted for every specific situation and region. For Clayoquot Sound, we suggest the criteria outlined in Table 1. The list of target species should be as comprehensive as possible and should be amended as new information becomes available. It should comprise species of all sizes and trophic levels, not only the large vertebrates.

Summarized, the strategy of target species has the following advantages for land management (Plachter 1991*b*,

Hansen et al. 1993, Altmoos 1997, Walter et al. 1998):

- abstract goals and rationales of conservation, such as ecosystem health or sustainability, become concrete and quantifiable;
- quantified goals can be monitored;
- savings of time and money compared with all-inclusive approaches;
- indirect contribution to the conservation of other species, habitats, and processes (umbrella effect);
- a hierarchical (in both a trophic and body size sense) array of target species allows the management and conservation of species and protected areas which considers all spatial scales;
- species remain the same in the planning time frames whereas ecosystems often change or fluctuate;
- some species rely on larger-scale connections and landscape functions that might be overlooked with other approaches;
- different executors of management plans will come to similar conclusions by using a quantitative, well-defined approach; and
- habitat degradation becomes quantifiable in terms of target species declines.

To manage a target species some basic information is required: the distribution and abundance (at least semiquantitatively); the ecology (feeding and breeding, habitat requirements, dispersal); and the basic life history attributes (survival, longevity, fecundity) of the species. Also important is information on the home range (where applicable) and threats. Note that any form of artificial enhancement must be excluded from the target species strategy. For more detailed discussion of the target species strategy, see Bahn (1998).

THE MARBLED MURRELET AS TARGET SPECIES

The marbled murrelet was one of several species and groups of species identified as important for land management decisions in Clayoquot Sound, British Columbia. Therefore, the British Columbia Ministry of Environment, Lands and Parks initiated an inventory of marbled murrelets there in 1995 (see Chatwin et al. 2000). The habitat suitability index put forward in this paper is based on the data and analyses from the 4 years of inventory (Bahn and Newsom 1999).

Marbled murrelet nesting habits are unique among members of the family Alcidae: these birds nest nearly exclusively in trees of old, coastal forests. The forests in which they breed are usually of high economic value and are often easily accessible from the ocean. Therefore, these forests have been and still are heavily exploited by the logging industry. Removal of nesting habitat (i.e., large trees), among other threats such as gill-netting and oil spills, has reduced the numbers of marbled murrelets in North America and caused this species to be recognized as threatened in both Canada and the United States (Rodway 1990, U.S. Fish and Wildlife Service 1992).

When evaluated according to the selection criteria for a target species (Table 1), the marbled murrelet is a promising candidate. It is highly threatened under current forest practices but is unlikely to become extinct if forestry practices improve, as promised by politicians and industry. It is indigenous to British Columbia and is highly dependent on old-growth forests as breeding habitat. Any alteration of breeding habitat is likely to decrease fecundity and long-term abundance of this bird. With its high requirements for breeding habitat, protecting the marbled murrelet would likely protect many other species dependent on old growth. The marbled murrelet is not known to have large-scale migrations although certain seasonal movements are known. In addition, the main cause of its "endangered" status is breeding habitat loss and not threats in other seasonal habitats. Furthermore, the marbled murrelet is thought to exhibit a certain breeding area fidelity, which means that it probably does not disperse quickly, although it is known to fly long distances in search of breeding habitat. Last, conservation efforts for murrelets have made this species well known and popular along the West Coast.

A drawback to selecting the marbled murrelet as a target species is that, despite the relatively large amounts of effort and money that have been put into marbled murrelet research, little is known about its basic life history. Furthermore, all demographic research is overshadowed by high unexplained variability in marbled murrelet inland and at-sea activity and much spatial movement on all time scales. Therefore, much more research into its life history and demography is needed.

Different species can serve to define and monitor different goals (Lambeck 1997). With its high spatial and structural demands on old-growth forests as breeding habitat, the marbled murrelet can be used as an indicator of both the quantity and quality of forestry practices. Preservation of sufficient old-growth and retention of a sufficient number of trees in logged areas (variable-retention system) will keep marbled murrelet numbers stable and facilitate recovery of breeding habitat (Scientific Panel for Sustainable Forest Practices in Clayoquot Sound 1995). In other words, the murrelet can be used to protect both habitat and other species dependent on old growth, as well as define minimum standards for forest practices. It is not useful in a strategy for the conservation of processes or connectivity in the landscape.

For the evaluation and adaptation of land management, continuous monitoring is indispensable (Plachter 1991a, Brown and Rowell 1997, Grumbine 1997). Murrelet population sizes can be measured by at-sea counts (e.g., Agler et al. 1998) or, on a watershed level, population levels can be indirectly monitored by radar counts of birds flying to their nesting sites (Burger 1997). For finer scale analyses of planning areas, audiovisual ground surveys (Ralph et al. 1994)

and random tree climbing for nests can be used in forests.

The first step is to monitor population levels before habitat disturbances. The second step is to establish limits of tolerance in population changes for individual watersheds (Bahn 1998). These limits cannot be scientifically derived but must be agreed upon by experts. Third, a detailed planning process involves the habitat suitability mappings described in the next section. Last, population levels after forestry impacts must be monitored and plans adapted according to the results.

HABITAT EVALUATION MODEL FOR THE MARBLED MURRELET BASED ON MAPPED INFORMATION

To consider the needs of marbled murrelets in management decisions, methods must be developed to evaluate potential marbled murrelet habitat in a standardized way. The habitat evaluation procedure (HEP) developed by the U.S. Fish and Wildlife Service (1980) is a standardized model for habitat evaluation which has been widely applied in North America (Gray et al. 1996). It is based on a habitat suitability index (HSI) (U.S. Fish and Wildlife Service 1981), which is calculated using habitat variables known or perceived to be important to a species.

The main objective for building this model for the marbled murrelet was to provide a management tool for the evaluation of coastal temperate rain forest as breeding habitat based on digital map information available. The model does not consider any other habitat requirements of the marbled murrelet (e.g., foraging habitat). Vegetation Resources Inventory (VRI) maps contain forest cover variables (e.g., tree height and age) that are linked to mapped polygons representing relatively homogenous areas (about 10 ha in size).

With field data from the inventory we identified the habitat variables most useful in characterizing marbled murrelet breeding habitat (i.e., density of nesting structures and forest structural characteristics). We also used information on marbled murrelet breeding habitat characteristics from the literature and biological reasoning to determine important habitat variables. We correlated these habitat variables with the forest cover variables from the VRI maps to identify useful forest cover variables and to determine their relationship to known marbled murrelet breeding habitat characteristics (Fig. 1). The identified variables were

- height (m) of the leading or the second leading species on the VRI maps (whichever was higher);
- age (yr) of the leading or the second leading species on the VRI maps (whichever was older);
- basal area (m^2/ha) of canopy and emergent trees from VRI maps;
- vertical complexity (1–4) of the forest canopy as defined by the VRI manual;
- canopy closure (%) of the tree crowns;

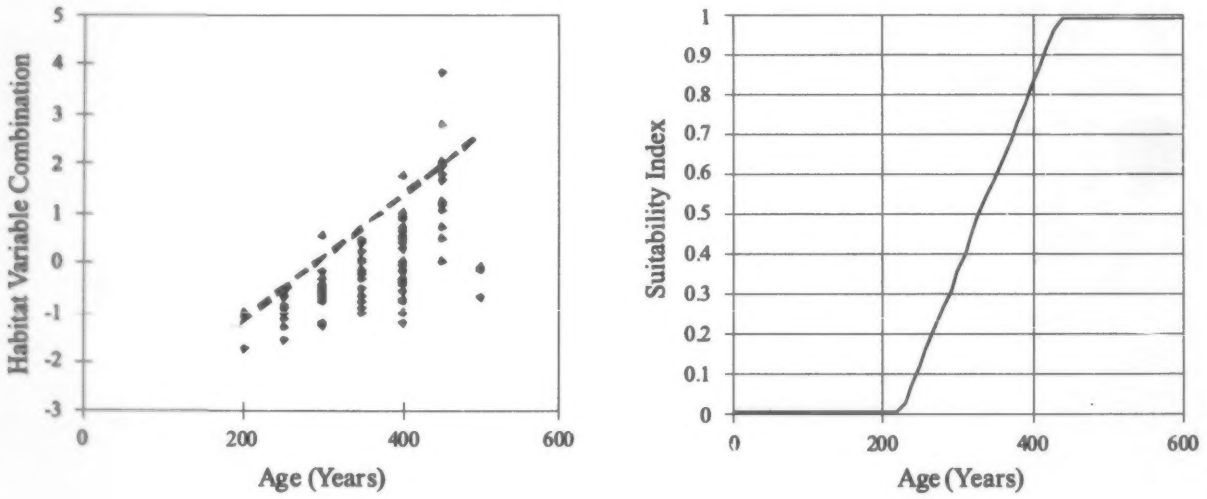

Figure 1. Construction of the marbled murrelet suitability index for the highest average age (yr) of the dominant and the second dominant tree species. This variable stems from a forest cover map (VRI). The graph on the left shows a scatter plot with age as independent and a principal component analysis combination of several habitat variables describing suitable marbled murrelet habitat as dependent variable. The habitat variables were derived from field data and literature. The dashed line is a least absolute deviation 90th quantile regression line. The right graph shows the translation of the left graph into a suitability index ranging between 0 and 1 (from Bahn and Newsom 1999).

- average distance of the polygon from the ocean; and
- average altitude of the polygon.

Another variable that is important for murrelet habitat quality is distance to the next major forest edge. However, we have been unable to make this information available in a digital form.

The next step was to construct evaluation graphs (called suitability indices) for the 7 selected forest cover variables. To determine their relationship to the habitat variables derived from field work, we combined the habitat variables in a principal component analysis (PCA) and made regressions between the PCA combination and the 7 forest cover variables. These regressions were the basis for suitability indices (SIs) which associate different values of the selected variable with an evaluation between 0 and 1. Fig. 1 shows 1 example of the derivation of an SI from a regression between the mapped forest cover variable and the habitat variable measured in the field. For a complete version, see Bahn and Newsom (1999).

The individual SIs for each forest cover variable were combined in a mathematical expression (weighted average) to form a single HSI. The mathematical expression was created according to our judgement of the importance and properties of the variables.

We are aware that, although the individual SIs were based on habitat features known to be important to marbled murrelets, many of the underlying assumptions are untested. For example, although we know that marbled murrelets use platforms for nesting, a definite relationship between density of platforms in a habitat and nesting density has never been established. More accurate relationships between the density of important habitat characteristics and nesting density can only be established by nesting density studies that ideally would include reproductive success. The ideal evaluation output from the model would be linearly related to nesting density and success of marbled murrelets.

Last, we produced maps showing the HSI scores of the individual polygons. For an informative map, we needed to group the polygons according to their HSI scores and to show them in different colours. In addition to the informative value, this grouping exercise is the point at which HSI scores must be translated into a verbal evaluation (i.e., habitat groupings from "excellent" to "unsuitable").

We suggest that this model could be applicable to within 50 km of the coast in the Coast and Mountains Ecoprovince. This range may be too wide and therefore must be reconsidered when the model is verified. This HSI model is a hypothesis which ought to be tested and refined as new results become available.

CONCLUSION

Talk is cheap. Concepts such as sustainability, ecosystem integrity, or ecosystem health are easily adopted into management vocabulary but remain inoperable until they are expressed quantitatively. The management of species and habitats at risk depends on measurable goals that can withstand economic pressure and can distinguish them from public relations rhetoric.

Although the target species strategy has some deficiencies, we favour it over ecosystem-based approaches to land management in areas where human use and conservation coincide. Shortcomings of the target species strategy are (1) that it has rarely been proven that the needs of 1 species adequately represent the needs of other species and (2) that insufficient information exists on potential target species. Nevertheless, we prefer it because it relies on species as entities that stay the same over time and can be counted. The success of this strategy depends on the selection of the target species and the tools used in their management.

The marbled murrelet is an obvious choice for a target species. Its high dependence on coastal old-growth forests makes its abundance and distribution a valuable indicator of the state of these forests.

The HSI we created for the marbled murrelet based on the research in Clayoquot Sound, British Columbia, could be an efficient tool for the management of this species on a large scale. The HSI bridges the gap between a purely scientific analysis of the habitat requirements of a species and the consideration of these requirements in management and planning. However, the HSI must still be verified. With the help of this tool the marbled murrelet is becoming an integral part of land management in Clayoquot Sound.

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Using Multi-Scale Inventory and Research to Conserve Marbled Murrelets in Clayoquot Sound, British Columbia

Trudy A. Chatwin
British Columbia Ministry of Environment, Lands and Parks
2080 Labieux Road, Nanaimo, BC, V9T 6J9, Canada
Trudy.Chatwin@gems1.gov.bc.ca

Lindsay E. Jones
British Columbia Ministry of Environment, Lands and Parks
202-4917 Pemberton Road,
Port Alberni, BC, V9Y 5J8, Canada

Alan E. Burger
Department of Biology, University of Victoria
Victoria, BC, V8W 3N5, Canada

Deanna Newsom
Central Westcoast Forest Society
Box 364, Ucluelet, BC, V0R 3A0, Canada

ABSTRACT

The marbled murrelet (*Brachyramphus marmoratus)* is a threatened species in Canada and is Red-listed in British Columbia. Clayoquot Sound on Vancouver Island, an area of high profile forest conflicts, supports one of the largest concentrations of breeding marbled murrelets in North America. In 1994, the Province of British Columbia appointed a Scientific Panel to make recommendations regarding forest management in Clayoquot Sound. One of the recommendations from the Clayoquot Scientific Panel was "to ensure that particular species known or suspected to be at risk are monitored and their habitats protected." Our goal is to provide quantitative information on the importance of Clayoquot Sound and habitats within Clayoquot Sound for marbled murrelets. We gathered data at a full range of spatial scales over 3 years using different inventory techniques. At the regional level over multiple years, we used boat surveys to estimate total numbers and marine distribution of the murrelets, allowing comparisons with other regions and estimations of population trends. At the watershed level, we used high frequency radar to count numbers of murrelets entering each watershed and correlated these numbers with large scale vegetation and topographic parameters such as amount of valley-bottom timber. At the forest stand level, we conducted standard audiovisual surveys to determine site nesting occupancy and to compare relative densities of marbled murrelet detections with a range of macro- and microhabitat parameters. We studied nesting density in an Ursus River valley-bottom site with high numbers of detections. Sea surveys showed a significant variation in year-to-year numbers and distribution of marbled murrelets. Using sea survey and radar data we estimate the population of marbled murrelets in Clayoquot Sound to be 6,000–8,000 birds. High radar detection numbers in a watershed correlated with size of watershed and amount of remaining valley-bottom timber. We found that "occupied" marbled murrelet detections were significantly correlated with large (both tall and large diameter) moss-covered coniferous trees in valley-bottom forest stands. A combination of ecosystem classification and 3 years of detection data was used in a geographic information system (GIS) mapping analysis to rank and map habitats of importance to marbled murrelets in the Ursus watershed. Tree-climbing of random trees with potential platforms demonstrated that in this undisturbed west coast watershed, marbled murrelets nest in very low densities. We continue to use this information in the planning processes that delineate protected habitat and management zones for marbled murrelets in Clayoquot Sound, and other coastal areas of British Columbia.

Key words: at-sea surveys, *Brachyramphus marmoratus*, Clayoquot Sound, conservation, inventory, marbled murrelet, multi-scale.

The marbled murrelet (*Brachyramphus marmoratus)*, a seabird of the family Alcidae, is listed as threatened by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) and is on the British Columbia Red List of species at risk (i.e., considered for legal designation as either extirpated, endangered, or threatened). Loss of nesting habitat in coastal old-growth forests is recognized as the principal threat to the species in British Columbia (Rodway et al. 1992) and elsewhere (Ralph et al. 1995). Clayoquot Sound has long been recognized as supporting large populations of

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

breeding marbled murrelets (Sealy and Carter 1984, Rodway et al. 1992, Burger 1995). Minimum estimates from sea surveys during the breeding season have ranged from 2,600 to 4,500 birds (Sealy and Carter 1984, Kelson et al 1995). There is evidence that the Clayoquot population is declining, and loss of nesting habitat in old growth is suggested as a cause of decline (Kelson et al. 1995).

Little was known about the inland distribution and habitat requirements of the murrelets within Clayoquot Sound when we began work in 1996. However, it was recognized that Clayoquot Sound's intact old-growth forests and abundance of near-shore feeding habitats were important to North American populations of marbled murrelets (Rodway et al. 1992).

Our 3-year inventory and research program began in May 1996 as part of the government's commitment to implement recommendations of the Scientific Panel for Sustainable Forest Practices in Clayoquot Sound (1995) for sustainable ecosystem management. The marbled murrelet inventory primarily addressed Recommendation 8.3.2: to "ensure that particular species known or suspected to be at-risk are monitored and their habitats protected" and Recommendation 7.16 which states: "at the watershed level map and designate reserves in which no harvesting will occur to protect red- and blue-listed species [and] late successional forest with forest interior conditions." In addition, we worked to address silvicultural recommendations 3.6 and 3.8: to assist in identifying retention areas with significant wildlife resource values and assist in selecting specific structures and patches to meet ecological objectives.

The program was designed to provide empirical data for ongoing monitoring of murrelets, both inland and at sea, and for assessing the effects of changes occurring in Clayoquot Sound. As well, our study provides tested methodologies which can be applied to conservation of marbled murrelet nesting habitat in coastal temperate rain forests throughout its range.

The objectives of our work were to:

1. estimate the total number of marbled murrelets found in Clayoquot Sound in summer;
2. estimate the numbers entering Clayoquot Sound watersheds during the breeding season, and assess the relative importance of each watershed as nesting habitat;
3. determine which parts of selected watersheds are most important to nesting murrelets;
4. identify critical macro- and microhabitat features and mapped ecosystem or forest vegetation units that provide high quality marbled murrelet nesting habitat;
5. determine nesting density of various ecological units, and describe nest stand, and nest tree characteristics;
6. assess the risks of predation to nesting marbled murrelets in a wide range of habitats;
7. assess how clearcut logging and forest fragmentation affect population densities, nest site selection, and predation risk in marbled murrelets;
8. monitor distribution and densities of marbled murrelets at sea including juvenile recruitment; and
9. assess seasonal and annual variations in all measurements, the factors causing these variations, and the effects of the variations on inventories and management decisions.

STUDY AREA

The study area (Fig. 1) is located in Clayoquot Sound on the west coast of Vancouver Island, British Columbia (latitude: 45°00'–49°30'N; longitude: 125°20'–126°30'W). The area is a maze of islands, inlets, and coastal temperate rain forest watersheds which are classified predominantly within the Coastal Western Hemlock (CWH) biogeoclimatic zone (Green and Klinka 1994) and the Windward Island Mountains Ecosection (Campbell et al. 1990). This is the rainiest zone in British Columbia and climate is characterized by cool summers and mild winters (Meidinger and Pojar 1991). The area we sampled ranged in elevation from 0 to 730 m, but mountains in the area range up to 1800 m. The vegetation of Clayoquot Sound is comprised of dense coniferous forest with trees ≤70 m in height and ≤2.5 m in diameter. The understory of these forests is characterized by a dense tangle of shrubs, deadfall trees, and moss. Clayoquot Sound contains several watersheds which have not been accessed for forest harvesting or other industrial development (Moore 1991). These intact watersheds form a substantial proportion of intact watersheds remaining on the west coast of North America.

METHODS

To meet objectives, we conducted several multi-scale inventories over each of the 3 years of the study. Each sub-study had a leader providing scientific direction. Detailed discussion of methodology can be found in Clayoquot Sound Marbled Murrelet Inventory for 1996 (Burger et al. 1997), 1997 (Chatwin et al. 1999), and 1998 (Bahn and Newsom in press).

Sea Surveys

In 1996, a quadrat survey of the entire Clayoquot Sound was conducted in a similar manner to sea surveys conducted in 1982 (Sealy and Carter 1984), 1992 and 1993 (Kelson and Mather in press) to compare distribution and abundance of marbled murrelets from year to year.

Standard sea surveys (Resources Inventory Committee 1995) were conducted in small boats from mid-May, every 2 weeks until at least mid-August. Fixed-width transects were used to estimate relative and absolute abundance (birds/km^2), spatial and temporal distribution, habitat use, and breeding chronology. The southern (Tofino) route (1996, 1997, and 1998) went from Tofino, south to Portland Point, around

Vargas Island, and back to Tofino. The second set of surveys (1997 and 1998) circumnavigated Flores Island. Both routes covered exposed coastal waters and sheltered channels.

Radar Inventory

Mobile high frequency marine surveillance radar units were used to sample up to 19 watersheds units in 1996, 1997, and 1998. As we were using a relatively new method for estimating populations of murrelets entering a watershed, we investigated the most appropriate sampling times, effects of weather, and various measures for estimating numbers. The scanner was generally set on a platform above the ground at the mouth of an inlet to provide a clear view across the expected flight path of murrelets to the watershed. We surveyed 2 hours before dawn and 1 hour afterwards and also at dusk. Only pre-sunrise counts were used to estimate mean and maximum counts per watershed. Comparisons among years and within seasons were done with paired *t*-tests. Maximum and mean counts of murrelets were compared with areas of logged, immature, mature/old growth, total forest, freshwater, other habitat (alpine, rock, ice), and total area of watershed.

Inland Audiovisual Surveys

Inland audiovisual surveys (Resources Inventory Committee 1995) were conducted at 177 stations throughout Clayoquot Sound over 3 years, during the breeding season (15 May to 15 July). In 1997, the most extensive and intensive year of

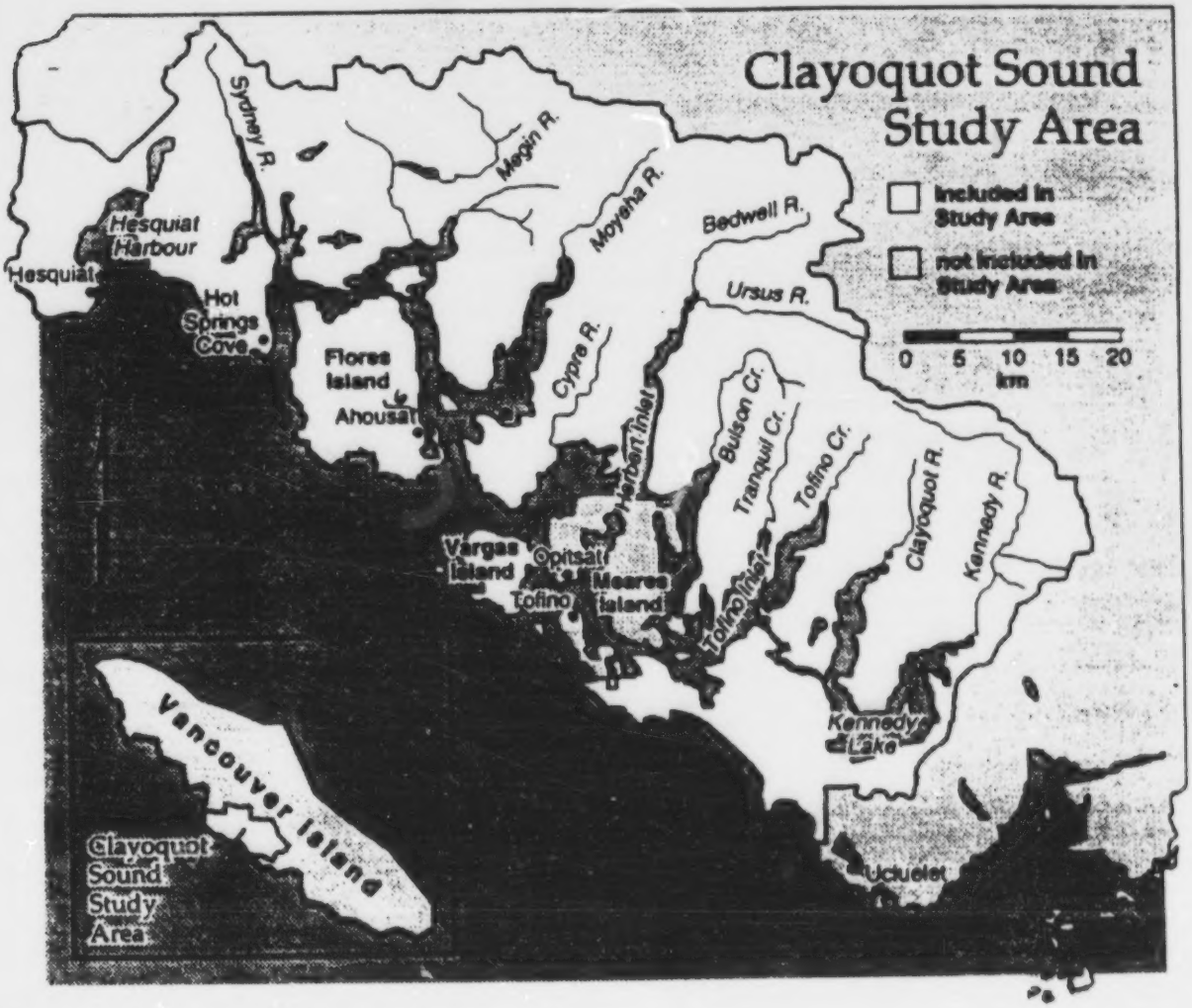

Figure 1. Clayoquot Sound study area.

sampling, stations were sampled 3 times over the season to control for variation in activity due to date. Surveys were conducted from 1 hour before to 1 hour after sunrise by trained observers. Murrelet behaviour and distance from observer, weather, number and species of potential predators, and standard landscape parameters were measured. Vegetation characteristics were measured in 30 X 30 m plots adjacent to each survey station. Ecosystem units were determined through examination of species composition, soils, and moisture regime. An in-depth statistical analysis of the relationship of occupied detections (detections indicative of nesting including landing in a tree, flights below canopy, and circling above a stand) to a suite of habitat characteristics was done using hierarchical regression models. Forest structural characteristics were compared with location in valley and subzone variants using 1-way analysis of variance (ANOVA).

Nesting Density Study

Twenty random plots were selected in valley-bottom and lower slope habitat (10 in each habitat type) in the Ursus Valley Special Management Area. The 12 coniferous trees with platforms closest to the plot centre were climbed and inspected for marbled murrelet nests. In 1998, 4 lower slope and 4 valley-bottom slope plots were climbed before weather closed in on climbers. An initial nesting density was calculated based on nests found over areas sampled in 1998, and observer bias in estimating platforms was calculated by comparing actual platform counts by climbers to platform estimates from the ground.

Intensive Monitoring of Marbled Murrelets in the Ursus Valley Special Management Area

During breeding season from 1995 (see Burger et al. 1995) through 1997, audiovisual surveys were conducted in combination with vegetation plots at 51 stations in a wide range of elevations, slopes, and ecosystems types across the Ursus Valley. We correlated murrelet activity with habitat at stations. Using the mapped ecosystem unit polygons (Clement 1995) and the murrelet activity data, we mapped and ranked polygons to indicate the most important areas for nesting murrelets. As well, we compared annual variation in murrelet activity at 12 survey stations that were surveyed every year between 1995 and 1998.

Testing Predictions on Nesting Habitat Quality

In 1998, we tested our ability to predict murrelet activity and habitat quality based on a single mapped forest characteristic. This analysis was based on polygons and habitat variables found on the Vegetation Resource Inventory maps (B.C. Ministry of Forests 1998). We chose to use the height of the dominant tree species as our predictive characteristic, as an analysis of 118 Ursus River vegetation plots showed that this mapped characteristic was most strongly correlated with habitat features thought to be important to murrelets, such as the amount of epiphyte cover, number of potential platforms, standard deviation of tree height (a measure of canopy complexity), and timber volume. We tested our predictions by simultaneously surveying murrelet activity in polygons in Bulson Creek and Flores Creek 6 that appeared to have good habitat and poor habitat (i.e., fell into different tree height categories). As well, we surveyed the vegetation characteristics associated with murrelet nesting (platforms, tree diameter, epiphyte cover, tree height, and species) in various mapped polygons considered to be good, moderate, or poor habitat (again based on tree height categories) in the Ursus watershed and looked for differences among the categories. Each mapped polygon in the Ursus watershed was evaluated using an index created from spatial and mapped attributes thought to be associated with nesting. (See Bahn [1998] for details of the Habitat Suitability Index.)

RESULTS

At-Sea Surveys

Marbled murrelets are abundant in Clayoquot Sound between mid-May and mid-July, then decline dramatically in the latter part of the summer. The quadrat census of the entire Clayoquot Sound (650 km^2) in 1982, 1992, 1993, and 1996 showed declines in total numbers of murrelets in the most populated inshore waters since 1982 (Fig. 2). The 1996 count of 4,036 murrelets was 22% less than the 1982 count of 5,178. Kelson and Mather (1999) attribute this observed decline to loss of nesting habitat from high rates of logging which occurred from 1982 to 1993. Mortality due to sports-

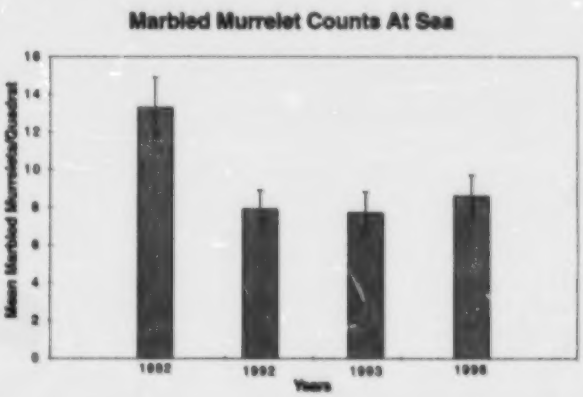

Figure 2. Means and standard errors of marbled murrelets per quadrat of at-sea surveys in the 4 years sampled. Means were significantly different (ANOVA, df = 3, $F = 5.22$, $P < 0.05$). Sheffe's pairwise comparisons indicated that numbers in 1982 were greater than in 1993.

Table 1. Means of all counts, means of maximum counts, and percentage of total counts at 19 watersheds in Clayoquot Sound in 1996–1998.

	Dawn surveys (Pre-sunrise count)				Dusk surveys			
Station	Overall mean	%	Mean of maximum	%	Overall mean	%	Mean of maximum	%
Atleo	87	1.9	118	2.1	56	4.5	69	3.9
Bedwell	361	7.8	441	8.0	139	11.2	303	17.2
Boat Basin	206	4.5	331	6.0	83	6.7	89	5.0
Bulson	326	7.1	412	7.4	96	7.8	158	9.0
Clayoquot River	385	8.3	494	8.9	52	4.2	75	4.2
Cow Bay	259	5.6	259	4.7	30	2.4	30	1.7
Cypre	42	0.9	63	1.1	31	2.5	40	2.3
Flores #6	108	2.3	120	2.2	32	2.6	45	2.5
Hesquiat Pt. Cr.	160	3.5	196	3.5	26	2.1	26	1.5
Kennedy	276	6.0	334	6.0	73	5.9	85	4.8
Meares (Lemmen's)	83	1.8	83	1.5	21	1.7	21	1.2
Megin	415	9.0	430	7.8	118	9.5	160	9.1
Moyeha	472	10.2	596	10.8	159	12.8	270	15.3
Pretty Girl	222	4.8	260	4.7	31	2.5	39	2.2
Sydney	180	3.9	225	4.1	44	3.6	52	2.9
Tofino Creek	212	4.6	213	3.8	36	2.9	46	2.6
Tranquil	172	3.7	268	4.8	47	3.8	63	3.6
Watta	628	13.6	650	11.7	155	12.5	182	10.3
Watta SE	30	0.7	45	0.8	9	0.7	12	0.7
Totals	4,624	100	5,537	100	1,238	100	1,765	100

fishing bycatch, the Nestucca oil spill, and an increase in water temperatures, which affects prey availability, are also mentioned as contributing factors to declines.

Season average for the Tofino transect ranges from 6.5 birds/km^2 in 1996, 8.9 birds/km^2 in 1997, to 9.1 birds/km^2 in 1998. Season average for the Flores Island transect ranges from 8.0 birds/km^2 in 1997 to 9.5 birds/km^2 in 1998 (Diggon et al. in press). The areas used by marbled murrelets tend to shift periodically through the season. However, the areas around islets off the southwest coasts of Wickaninnish Island, Vargas Island, and Flores Island, as well as Calmus Passage, had high densities of marbled murrelets in all years. In 1997, the juvenile percentage was 2% on the Tofino transect and 3% on the Flores transect. In 1998, the juvenile percentage was 5% for both transects. Further monitoring is required to assess long-term population trends.

Watershed Radar Counts

Dawn counts of murrelets were consistently higher than dusk counts, but both counts produced a consistent ranking of watersheds (Table 1). In most dawn surveys there was a unimodal pre-sunrise pulse of incoming murrelets, but a few surveys found a second post-sunrise pulse. We therefore used pre-sunrise counts when making estimates of mean and maximum number of murrelets entering a watershed. We found no effect of date on dawn counts between 14 May and 16 July and this core period was used to compare watersheds. The radar inventory showed that over 4,600 marbled murrelets (mean of the sum of annual mean counts per station) and perhaps as many as 5,500 (sum of mean of annual maximum counts) were using the 19 watersheds. This count includes nonbreeding murrelets. Considering the areas not sampled with radar, the regional population of Clayoquot Sound is estimated to be at least 6,000–8,000 breeding and nonbreeding marbled murrelets.

About 80% of the marbled murrelets counted were at 7 watersheds: Moyeha, Watta, Megin, Clayoquot, Kennedy, Bedwell-Ursus, and Bulson rivers, as well as Boat Basin, and Flores Island. Numbers of murrelets per watershed were positively correlated with the area of remaining mature and old-growth forest (>140 years old).

Stations sampled over 2 years showed no significant difference between the years in total maximum or mean counts. However, there were some changes among watersheds between years. This indicates that while the total number of marbled murrelets remained constant in Clayoquot Sound between 1996 and 1998, there was movement of murrelets between watersheds in the various years. Therefore, multiyear radar sampling is required to correctly assess the relative importance of watersheds.

Inland Inventory: Marbled Murrelet Activity, and Structural Characteristics of Old-Growth Forests

Numbers of occupied, within 50 m, and sub-canopy detections, all considered to be indications of marbled murrelet nesting , were positively related to forest structural characteristics known to be important to nesting murrelets (Hamer and Nelson 1995), especially density of trees with platforms (a branch over 18 cm in diameter including branch and moss that could be used by a nesting marbled murrelet), density of trees over 80 cm diameter at breast height (dbh), and mean diameter of all trees (Rodway and Regehr 1999*b*).

Numbers of occupied, within 50 m, and sub-canopy detections were greater at valley-bottom stations located at stream channels. It could not be determined if this activity was associated with local habitat use or with flight corridors (Rodway and Regehr 1999*a*) used to access other parts of the valley. However, most structural characteristics associated with known murrelet sites, such as high mean tree diameter, greater epiphyte cover, and high density of large trees (trees >80 cm diameter), were more prevalent in valley-bottom than slope habitats. Density of trees with potential nesting platforms decreased with distance from major streams.

Biogeoclimatic subzone variant CWHvm1 had higher numbers of detections than other subzones (CWHvm2 and CWHmm1). Within this subzone variant the High Bench Floodplain Sitka Spruce/Salmonberry site series rated highest for structural characteristics important to nesting murrelets. Low numbers of detections at stations close to the ocean edge at outer coast CWHvh1 habitat, lower densities of trees with potential nesting platforms at the ocean edge, and elevated predator levels at the ocean edge all suggest that perimeter forest along the ocean would be poorer habitat for murrelets. Vegetation site series with poorer nutrient and moisture regimes generally rated poorer for forest structural characteristics important to nesting murrelets. However, differences in occupied, within 50 m, and sub-canopy detections between vegetation site series were generally not discernible.

Numbers of detections tended to be highest in Bulson, Flores Creek 6, Ursus, Hesquiaht Point, and Sydney watersheds. Forest stands fragmented by logging had significantly lower detection rates than unfragmented forests, and the murrelet predators, Steller's jays and northwestern crows, were more frequent in areas fragmented by logging.

Numbers of detections were highly variable and were influenced by weather, date, opening size, and the placement of survey stations. The general association with forest characteristics and habitat use by marbled murrelets suggests that structural characteristics may be a more useful method than inland surveys for differentiating and prioritizing murrelet breeding habitat at stand scales.

Nesting Density Study

Although the nesting density study is not complete, 93 coniferous trees with platforms were climbed in 4 valley-bottom and 4 lower slope plots. The total area of valley-bottom plots and lower slope plots climbed was 130,000 m^2. In this area, 3 marbled murrelet nests were found; 2 were in large diameter western redcedars in lower slope plots and 1 was in a large diameter Sitka spruce in a valley-bottom plot. We found that actual platform counts made by tree climbers were significantly higher than ground observer platform counts.

The calculated nesting density for lower slope and valley bottom habitat is 0.18 (SE = 1.39) nests/ha/year. Applying this density to the areas of "excellent" habitat derived from the habitat suitability evaluation in the Ursus Valley (Newsom and Bahn 1999) and comparing the 355 nests with the radar estimate of 446 marbled murrelets entering the Ursus watershed (Burger in press), we recognize that the nesting density mean is probably high overall. However, this is the first unbiased estimate of nesting density in an unfragmented forest and it is encouraging that this estimate is not entirely inconsistent with radar estimates. Nesting density estimates are used to correlate the amount of suitable habitat with number of nesting murrelets and make more informed management decisions.

Intensive Monitoring of Marbled Murrelets in the Ursus Valley Special Management Area

Average total, occupied, and sub-canopy activity levels in the Ursus Valley fluctuated between 1995 and 1998. Generally, activity levels were high in 1995 and 1997 and lower in 1996 and 1998. These annual differences were significant ($P < 0.05$) for total detections and log-transformed occupied detections. However, individual station means for all measures of activity were not significantly different for any station. The relative importance of each station and geographical area within the Ursus Valley varied from year to year. For example, survey stations within the Thunderbird Creek tributary accounted for over 50% of total occupied activity in 1997, but less than 15% in 1998. The variability in relative amounts of occupied activity at both individual survey stations and areas may indicate that marbled murrelets shift their breeding focus among years (Bahn and Newsom in press). Shifting nesting areas could be a predator avoidance strategy, but also emphasizes the need for caution in determining critical habitat areas based on 1 season's activity monitoring, and the possibility that marbled murrelets may need large areas to ensure breeding success.

Testing Predictions on Nesting Habitat Quality

The number of occupied and sub-canopy detections were significantly higher in polygons where the Vegetation Resource Inventory maps showed average tree height was >35 m than in those mapped polygons where average tree

height was <26 m. Abundance of habitat variables predicted to be important for marbled murrelet nesting such as the density of trees with platforms and epiphyte abundance correlated with height of dominant trees on the Vegetation Resource Inventory maps. The values for these variables were significantly different among tree height classes; as expected, polygons with leading species over 35 m high had the highest number of platforms and thick epiphytes, followed by polygons with leading trees 26–35 m high, and finally polygons with leading trees <26 m high. In general, the height category 26–35 m contained variable nesting habitat for murrelets. The vegetation structure data from plots was used to construct a habitat suitability model for use with GIS polygon data from the Vegetation Resource Inventory (Bahn and Newsom in press). The model calculates a suitability index for each polygon based on tree age, height, basal area, vertical complexity, canopy closure, distance to the sea, and average altitude of the polygon. Nesting habitat suitability was then applied as 4 classes (unsuitable, sub-optimal, important-good, important-high) to the Ursus watershed (Fig. 3) and other areas in Clayoquot Sound.

CONCLUSIONS

1. Marbled murrelets are abundant in Clayoquot Sound in May through July. Although numbers of birds at-sea appeared to decline from 1982 to 1992, numbers remained relatively constant from 1996 to 1998.
2. We advanced the use of radar as a method of assessing populations of marbled murrelets at the watershed scale. The population of marbled murrelets in Clayoquot Sound is estimated to be 6,000–8,000 birds. Large and intact watersheds containing mature and old-growth forests support most of this population.
3. Correlation of audiovisual surveys and vegetation plots confirmed the association of marbled murrelet nesting

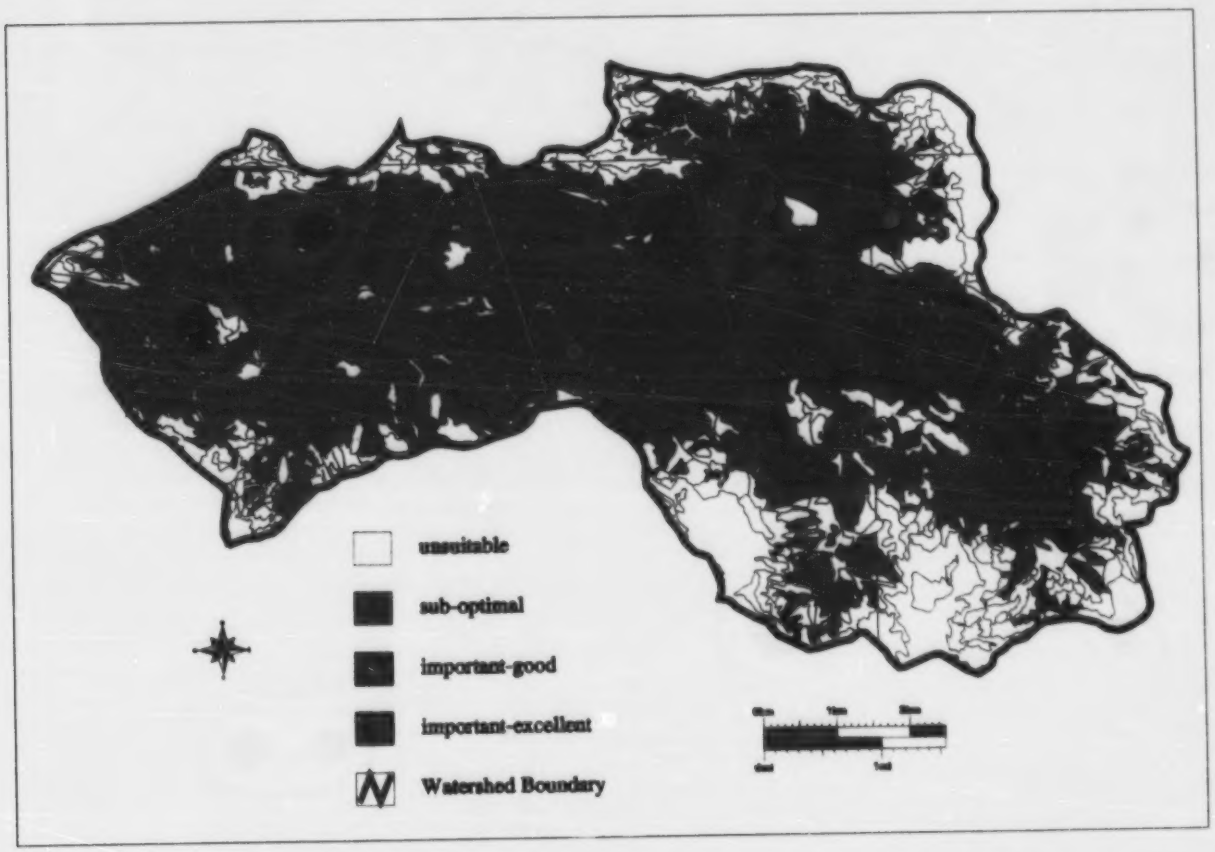

Figure 3. Marbled murrelet habitat suitability evaluation in the Ursus Valley, based on 5 forest characteristics from Vegetation Resource Inventory maps and 2 spatial characteristics. Characteristics considered for each polygon are: tree age, tree height, vertical complexity, canopy closure, distance from ocean, and elevation. Map produced by V. Bahn, D. Newsom, and K. Hodgson.

with tall, large diameter conifers with large mossy boughs. Our research suggests that evaluating a combination of mapped vegetation attributes and structures associated with marbled murrelet nesting may be the most effective and appropriate way of prioritizing forest nesting habitat for marbled murrelets.

4. Although the nesting density study is not finished, we provided the first estimate of nesting density in "excellent" habitat in an unfragmented watershed. Nesting density estimates provide resource managers with vital information for decision-making.
5. Through multiyear and multi-scale inventory we confirmed population estimates and checked the determinations of critical habitat. There is now a well-established baseline inventory of marbled murrelet distribution and abundance against which to measure the effects of future management in Clayoquot Sound.

ACKNOWLEDGEMENTS

Without Forest Renewal BC's financial support, this study would not have taken place. The Ahoushat, Hesquiaht, and Tla-oqui-aht First Nations supported our work and gave permission to work on their lands. The Central Westcoast Forest Society and British Columbia Conservation Foundation hired most of the workers, emphasizing loyal employees, local recruits, and the best experts in the field of Canadian marbled murrelet research: Dr. A. Burger, M. Rodway, H. Regehr, B. Beasley, and V. Bahn. V. Bahn was an especially keen worker as he was conducting his Master's research on conservation and nesting habitat suitability in the Ursus Valley. All of the field workers and volunteers, including L. Blight, E. Grundmann, K. Safford, N. Denis, G. Carrow, A. Lawrence, A. Burger, V. Bahn, D. Newsom, B. Foster, V. Bourdages, C. Bryden, R. Schoop, A. Daley, J. Johansen, M. Rodway, and H. Regehr, braved very wet summers in the second-wettest place on earth, flooding floodplains, rising rivers, thick bush, hordes of no-see-ums, and 0400 starts to diligently collect the data that went into the reports. K. Jordan, J. Boyle, and S. Hughes climbed the trees in the nesting density study. The pilots at Tofino Air, Harbour City, Long Beach and Vancouver Island Helicopters, J. Tom and A. Tom of Opitsaht, and the crew from Crystal Spirit Adventures transported us safely to remote locations in Clayoquot Sound. J. Clelland of the Clayoquot Biosphere Project tracked everyone's whereabouts for safety. M. Rodway and H. Regehr organized years of the inland survey data for S. Christensen to enter into the Ministry of Environment's Oracle database and for K. Hodgson to enter into the Ministry's GIS. M. Mather helped review and organize much of the work. This was a collaborative and cooperative project—everyone (even those mistakenly not mentioned) contributed to gain understanding of the life and requirements of the mysterious marbled murrelet.

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Marbled Murrelet Research in Desolation Sound, British Columbia

Cindy L. Hull[1]
Wildlife Ecology Chair, Department Biological Sciences, Simon Fraser University
8888 University Drive, Burnaby, BC, V5A 1S6, Canada

ABSTRACT

Research on marbled murrelets (*Brachyramphus marmoratus)*, based primarily at Desolation Sound, British Columbia, was started in 1991. The research, a collaborative effort of the Wildlife Ecology Chair at Simon Fraser University, the Canadian Wildlife Service, Natural Sciences and Engineering Research Council, and the British Columbia Ministry of Forests, aims to understand the population dynamics and marine and forest habitat requirements of this threatened species. By the end of the 1998 breeding season, 1,269 murrelets (including 115 recaptures and 55 recently fledged juveniles) had been captured and banded at Desolation Sound, providing information on survival, longevity, and site fidelity, and allowing annual and sexual morphometric comparisons. A molecular sexing technique, and hormonal assays that indicate breeding status was also developed. Radiotelemetry has been used to locate nest sites and to determine use of the marine environment. Twenty-three nest sites were located in 1998 using this technique. Murrelet activity has also been monitored and the nesting stands and trees described in the forest interior and at forest edge sites in the Bunster Range. All these data are being analyzed to determine the demography and habitat requirements of marbled murrelets in Desolation Sound.

Key words: *Brachyramphus marmoratus*, capture methods, Desolation Sound, endocrinology, marbled murrelet, mist-net, nesting, radiotelemetry.

Marbled murrelets (*Brachyramphus marmoratus*) are small alcids found along the Pacific coast of North America from northern California to Alaska. In British Columbia, they nest solitarily or in loose associations in large trees of old-growth (>100 years old) forest (Hamer and Nelson 1995). They can nest up to 60 km inland, but require the marine habitat for their food supply, predominantly Pacific sand lance (*Ammodytes hexapterus*), which they forage on close in-shore (Carter and Sealy 1990).

Populations of marbled murrelets are believed to be declining due to the removal and fragmentation of old-growth forest, bycatch in gill net fisheries, recreational usage in the marine habitat, urban development, and oil spills (Carter and Sealy 1984, Nelson et al. 1992, Paton 1994, Carter and Kuletz 1995, Fry 1995). These anthropogenic factors combine with what appears to be a low fecundity and late maturity (Nelson and Hamer 1995) to cause concern for the survival of the species. Marbled murrelets are classified as threatened or endangered by the U.S. Fish and Wildlife Service in Washington, Oregon, and California (Federal Register of the USA), and are a nationally threatened species in Canada (Committee on the Status of Endangered Wildlife in Canada), with Red-listing in British Columbia . While the population is believed to be threatened by logging practices, few empirical data are available on the status of the population or the requirements needed to maintain a healthy population. Effective management and recovery of an endangered species requires information on its population change and a correction in the factors that limit population growth. However, without long-term data, this has been difficult to achieve with marbled murrelets (Beissenger 1995). Still little is known about the demography of this species, survivorship for birds at any age, age at first breeding, or the proportion of adults that breed (Beissenger 1995).

The threatened status of marbled murrelets in British Columbia, and their vulnerability to changes in forest ecosystems from forestry practices, prompted a multidisciplinary research program in Desolation Sound (50°04'N, 124°42'W). The program, started in 1991, is a collaboration of the Canadian Wildlife Service (CWS)/Natural Sciences and Engineering Research Council (NSERC) Chair of Wildlife Ecology at Simon Fraser University (SFU), the British Columbia Ministry of Forests (MOF), and the CWS. It has been directed by Andrew Derocher and Louise Waterhouse (MOF); Gary Kaiser, Sean Boyd and Kathy Martin (CWS); and Tony Williams and Fred Cooke (SFU). Other personnel are involved in the program, including a postdoctoral fellow (author), and graduate and summer students.

This collaborative project has 2 main objectives: 1) to

[1] Present address: School of Zoology, University of Tasmania, GPO Box 252-05, Hobart 7001, Tasmania, Australia
cl_hull@postoffice.utas.edu.au

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb ,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

understand the population dynamics of marbled murrelets in Desolation Sound, in particular determine if the population is increasing, decreasing, or stable; and 2) to understand the marine and forest habitat requirements of the species. To achieve these aims, the program is undertaking:

1. Capture–Mark–Recapture (CMR) studies to ascertain movements, longevity, survival rates, and population trends of marbled murrelets. These data are essential for understanding the population dynamics of the species. They also provide valuable information on morphometrics and allow an assessment of the variability within this population and its discreteness from other populations. An assessment of the movements of marbled murrelets using marine radar assists with data on seasonal movements and population trends, and with other aspects of the program, breeding chronology.
2. Endocrinology studies, to determine sex and breeding status of marbled murrelets, data that cannot be readily obtained on this species in other ways. Information about the breeding status of marbled murrelets will eventually provide data on age at first breeding which will assist with determining the status of the population.
3. Habitat usage in both the marine and forest environments, using observations of colour-marked individuals and radiotracking. Radiotracking allows nest sites to be located, and an assessment of the proportion of birds that breed amongst the sample tracked.
4. Nest site and habitat descriptions, by conducting systematic ground searches and tree-climbing in old-growth forest stands.
5. Surveys of marbled murrelet activity in the forest for information on site fidelity, and seasonal changes in murrelet activity and abundance to allow comparisons between nests found in clearcut, forest edge, and forest interior sites.

These various techniques provide multiple estimates of population size and trends, and the biases inherent in each methodology can be assessed.

Research on marbled murrelets is difficult due to their secretive habits—the first active nest was not discovered in British Columbia until 1993 (Jones 1993), and due to their vulnerability to disturbance at or near the nest (Long and Ralph 1998). Therefore, many of the usual seabird protocols cannot be used on this species. To address the objectives of the program, a number of techniques have been tested and developed at Desolation Sound.

This paper describes the methodologies used for studying marbled murrelets at Desolation Sound, a summary of findings, and research planned for upcoming years. The results of forest habitat studies will be discussed elsewhere (Manley and Blackburn 2000), only a preliminary description will be given here.

METHODS

Study Site and Capture Methods

Desolation Sound and the adjoining Malaspina, Okeover, Lancelot, and Theodosia inlets were selected as the study site due to the many marbled murrelets known to be in the area and to previous work at this site (Kaiser et al. 1991, Mahon et al. 1992). Theodosia Inlet was selected as the major capture site as many murrelets are known to move through this inlet between the forest and Malaspina and Lancelot inlets during May–July (Kaiser et al. 1991, Mahon et al. 1992). Theodosia Inlet is narrow (200 m between shoreline trees), allowing birds to be captured with mist-nets (Derocher et al. 1996).

A floating mist-net system (Kaiser et al. 1991) was developed during this program which has allowed the capture of many marbled murrelets in Desolation Sound. Birds were intercepted as they flew low over the water to and from the forest. Mist-netting was carried out using a 2 mist-net array. Each array contained 4 mist-nets, attached to poles which are inserted into flotation devices. Using a pulley system, nets were pulled across the inlet by personnel in small boats. One array was set southward and the other northward, effectively blocking Theodosia Inlet with a 30% overlap. The net systems were separated by about 100 m to allow the safe passage of boats. Mist-nets are typically operated daily for 2–3 hours around dusk and again around dawn. Nets were generally left open for 30 minutes after the capture of the last bird. Birds were removed from the net by 1 person as a second person positioned an inflatable boat beside the capture site (see Drever et al. 1998, Lougheed et al. 1998*a* for further details).

During 1997 a night-lighting technique was developed (modified from Whitworth 1997) to capture on the open waters of Desolation Sound at night. Night-lighting allowed the capture of birds earlier in the breeding season than mist-netting, and for the first time at this site, the capture of juveniles. Personnel in inflatable boats used spotlights to scan for murrelets, using the birds' eye colour and shape as an identifier. Once a murrelet was observed, it was approached at minimum speed, with the spotlight used to dazzle the bird. When the bird was at close range it was scooped into the boat using a landing net (see Lougheed et al. 1998*b* for further details).

Capture–Mark–Recapture

Murrelets captured by either mist-netting or night-lighting were banded with size 3 stainless steel U.S. Fish and Wildlife Service/CWS bands. Since 1994 the following body measurements were recorded, using calipers except where stated (based on Prater et al. 1977): flattened wing chord (measured from the wrist joint to the tip of the longest primary using a metal ruler); exposed culmen; bill width (measured just anterior of the nares); bill height (measured just anterior to the

edge of the cuticle); tarsus (from the end of the tarsus at the inter-tarsal joint to the foot scale at which the toes emerge); brood patch area; brood patch condition as an indication of breeding (following Sealy 1974); and mass using a Pesola scale.

Numbers of marbled murrelets moving between the forest and inlets were monitored using marine radar. A Raytheon Model 3604 radar, fitted with a 1.1-m end-fed slotted wave guide array antenna was used to survey Theodosia Inlet. Its range was 0.25 nautical miles, with murrelets the most common birds in the inlet and easily identified by their flight behaviour. The radar was linked with video equipment, and operated concurrently with mist-netting. This system has been used since 1996.

Endocrinology

A sample of blood was taken from a pin-prick in the tarsal vein of birds to determine their sex using a molecular sexing technique. The techniques had been modified from Griffiths et al. (1996) and adapted for marbled murrelets (Vanderkist et al. 1999). Biochemical and hormonal analyses were also performed on blood plasma samples from some birds to find indirect evidence of reproduction. Plasma levels of egg-yolk precursors were analyzed to detect fecund females and provide information on breeding chronology. Plasma levels of the steroid hormone testosterone were measured to detect periods of territoriality and aggression, typically associated with the beginning of reproduction. The reliability of these precursors had been validated by comparison with the closely related Cassin's auklet (*Ptychoramphus aleuticus*) where breeding status could be more easily inferred due to more precise breeding chronology data (Vanderkist 1999).

Habitat Usage

Use of the marine and forest habitat was assessed using a variety of techniques. To determine use of the marine environment and movement of individuals, murrelets were individually colour marked, using methods including turkey feathers, wing-tags, and nasal discs. The nasal discs, specifically designed for marbled murrelets, are a small circle (5 mm diameter) of 0.5-mm thick "Darvik" ultra-violet resistant plastic. The discs were attached on either side of the bill with a short piece of plastic fishing line melted to a small bead at each end and passed through the nares. The discs were designed to remain attached for 3 years. Wing tags were feather-shaped (3 x 1.5 cm) pieces of "Tyvek," a durable and flexible water-resistant paper, also specifically shaped for marbled murrelets. The attachment technique was modified from Curtis et al. (1983), with tags wrapped around the ulna on each wing and glued to themselves with a temporary adhesive. Wing tags were designed to fall off during molt, 2–3 months after attachment.

Radiotracking was also used to locate nests in the forest, and to monitor murrelet use of the marine environment, and movements of individuals. Two-stage BD - 2G (Holohil Systems Ltd, Woodlawn, Ontario) radio transmitters were attached to a sample of marbled murrelets. The transmitters measured 20 x 8 x 7 mm and had a flexible 150-mm antenna at the posterior end of the device, angled at 30° from the base. The devices weighed 1.45 g, which is approximately 0.73% average mass of marbled murrelets, with a cross-sectional area of 56 mm^2, or approximately 2.5% of the cross-sectional area of an adult marbled murrelet. The attachment techniques have varied between years. In 1994, transmitters were attached with a collar of nylon-covered elastic threaded through a channel in the transmitter body, with the ends of the elastic joined by a corrodable iron staple. The transmitter was held between the scapulars with a drop of cyanoacrylic glue. In 1995, transmitters were sutured and glued using a technique adapted from Wheeler (1991). During 1996, they were attached with "Bird Adhesive" glue (Titan Corporation, USA) between the scapulars. During 1997, transmitters were attached with epoxy to a small piece of fiberglass insect screen to increase the surface area of attachment. The screen was cut into a "wing" shape to conform to the bird's feathers. In 1998, subcutaneous anchors were successfully used. In all years, transmitters were designed to remain on birds for ≥6 weeks, when they fall off.

Radiotracking was conducted using Telonics TR-2 receivers coupled with Ts-1 programmable scanners and H-antennae. On the water, tracking was conducted from an inflatable boat using a 3-element directional antenna mounted on a 2.5-m swivelling aluminum pole. Aerial telemetry is carried out from both a Bell 206 JetRanger helicopter and a Cessna 172 fixed-wing aircraft. For helicopter tracking, a dual antenna and receiver system with 2 telemetry personnel was used to maximize detection probability. Four H-antennae, 2 on each side of the aircraft, were mounted on wooden poles strapped to the helicopter struts. Telemetry personnel had a switch box connected by 5-m coaxial cables to 2 antennae, one on each side of the helicopter. Switch boxes allowed direction to be determined by moving back and forth between the 2 antenna arrays. Flights were conducted at 200–3,000 m, starting at high altitudes for maximum range to determine presence and then lower, circling to pinpoint signals. A global positioning system (GPS) was used to determine the location of signals.

The movement of individuals and use of the marine habitat were also observed during at-sea surveys in Desolation Sound and surrounding inlets each morning and afternoon in inflatable boats using strip transects through 71 defined polygons. Sea surface temperature and salinity were recorded in each polygon. The total numbers of murrelets were counted, and group sizes noted. Fish-holding behaviour of murrelets was also recorded, because it indicates active breeding.

An assessment of the effects of different marking techniques and attachment of radiotransmitters on the maintenance,

social and diving behaviour of marbled murrelets was conducted during the 1996 season (Lougheed et al. submitted). The results of this study are being used to determine the marking procedures in future years of the program.

Nest Site and Habitat Descriptions

To determine marbled murrelet use of forest types, 36 stands were selected from accessible forests in the Bunster Range using 1:20,000 forest cover maps. Selected stands included low and high elevation, second- and old-growth forests, in 3 height classes.

Each stand was surveyed for murrelet activity 2–4 times during a breeding season. The forest and vegetation characteristics at the 36 stands were assessed using randomly chosen vegetation plots and transects. We measured the following variables: diameter at breast height, height, canopy lift, strata, vigour, the number and characteristics of all potential nesting platforms of all trees within a 25-m radius of a centre canopy tree, latitude, longitude, elevation, distance to nearest body of water, signs or presence of potential predators, and percent cover of indicator species. Any signs of murrelet presence (feathers, eggshells, prey items) were collected.

Surveys were also conducted once a week at 2 monitoring sites. Monitoring sites were located on the main logging road in the Bunster Range and were separated by 1 km. Nest trees were also searched during dawn surveys, egg-shell transects, and tree climbing. Nest trees and nest sites were documented following Ralph et al. (1994). All vegetation characteristics were measured using vegetation plots, with the nest tree as the centre canopy tree. Nest sites were evaluated at 4 scales: element, patch, stand, and landscape. At the element scale, nest site selectivity was evaluated by comparing nest trees with others available within a 25-m radius around the nest tree. At the patch scale, habitat variables from vegetation plots at the nest tree were compared with variables from a randomly chosen plot located within 200 m of the nest tree.

Pilot Studies at Other Sites

Pilot capture and banding studies were conducted at the Queen Charlotte Islands (52°46'N, 131°57'W) in 1996, and at Mussel Inlet (52°51'N, 128°10'W) in 1997. These sites were studied to compare data from birds at less disturbed sites than those at Desolation Sound, and to explore the male-biased sex ratio found at Theodosia Inlet.

RESULTS

The program at Desolation Sound will shortly begin its ninth year. Work during 1991 and 1993 (no work was conducted during 1992 due to funding constraints) was primarily pilot studies with the development of various techniques. In 1994, SFU joined the program, and banding effort was increased and the forestry component initiated. Many novel techniques have been developed during the program and are now in place, allowing the collection of various ecological and biological data on marbled murrelets at this site. Much of the data is still being analyzed, hence many of the following results presented are preliminary.

Study Site and Capture Methods

The 2 capture techniques developed and refined during this study after the 1998 breeding season have allowed the capture of 1,269 marbled murrelets. Night-lighting has allowed the capture of murrelets earlier in the breeding season, and for the first time at this site, recently fledged juveniles. These 2 capture techniques have meant that data can be collected for numerous murrelets and we no longer rely on dead birds (Sealy 1974).

Capture–Mark–Recapture

Following 5 years of intensive banding we can now estimate survival at 85% for marbled murrelets at Desolation Sound, and believe the population is approximately 5,000 birds (Lougheed et al. 1998*b*). The intensive banding program at Desolation Sound has allowed an update on the longevity of this species. An adult bird banded in 1991 was recaptured during 1998, indicating a longevity of 8 years. Some 150 murrelets have been recaptured during this program, for the first time indicating some site fidelity in this species. The recapture rate of 10% suggests either that there is a larger population than previously thought at Desolation Sound, or that murrelets have learned to avoid the mist-nets. The latter is unlikely as the effort required to capture marbled murrelets at Desolation Sound has not significantly increased across the years (although it does show annual variability; L. Lougheed unpubl. data).

Data from nests found during radiotracking (see following) have allowed a preliminary estimate of breeding success of 0.4, and an estimated fecundity of 0.3 for marbled murrelets at Desolation Sound. These preliminary assessments allow an estimate of the status of the population at this site. The results suggest that the population may be declining by 2.2–4.1% per year (Lougheed et al. 1998*b*, Table 1). However, the confidence limits on this estimate are large, and only with long-term studies and larger sample sizes can the confidence limits be reduced and estimates improved.

The capture and banding program has also provided other incidental information on the behaviour of marbled murrelets. More murrelets have been captured at dusk than dawn at this site, which indicates their diurnal activity and probably reflects an increase in feeding trips to chicks at dusk (Nelson and Hamer 1995, Burger 1997). Most murrelets caught in mist-nets are flying out of the inlet, which has also been confirmed by radar counts, with 94% of murrelets flying out of the inlet. Peak movement through Theodosia Inlet is

Table 1. Estimates of the parameters used to calculate the population status of Marbled Murrelets in Desolation Sound.

Variable	Mean ± standard error	95% confidence intervals
Survival	0.848±0.085	0.60-0.95
Predicted[a]	0.845	0.84-0.88
Breeding success rate	0.70	0.52-0.82
Hatching success rate	0.68	0.46-0.84
Fledging success rate	0.69	0.43-0.86
Nest success rate (hatching x fledging)	0.41	0.23-0.68
Fecundity (breeding success x hatching success x fledging success)	0.29±0.09	0.16-0.46
Population change	0.964±0.08[b]	0.804-1.24

[a] Beissinger 1995
[b] Decline of 3.6%/yr.

between 2100–0000 and 0500–0800 (L. Lougheed et al. 1998*b*). This result indicates the use of different flight paths by birds departing and returning to the forest; thus, mist-netting does not randomly sample the population at Desolation Sound, important information when assumptions are made about the entire population.

Morphometric data collected on marbled murrelets from Desolation Sound and Mussel Inlet from 1994 to 1997 have been analyzed and details can be obtained from Hull et al. (submitted). No differences were found in the morphometrics of murrelets captured during different years or times of day of the study, or in birds captured using mist-netting or night-lighting. However, significant differences in culmen, wing chord, and tarsus were found between birds from Mussel Inlet and Desolation Sound, indicating discrete populations.

The degree of sexual dimorphism in this species is small (98%) and a discriminant function analysis revealed only a 70% success rate in allocating birds to sex using morphometric data (Hull et al. submitted). Therefore, the molecular sexing technique of Vanderkist et al. (1999) (see following) must be used to avoid errors when sexing birds. We found significant differences in measurements between observers. Wing chord had the highest percent measurement error (66.8) between observers, and tarsus, the lowest (36.8). These differences were found although all personnel had been trained in measurement protocols. Residuals were effective in removing most observer effects and allowed comparisons within and between sites (Hull et al. submitted). We will continue to take all morphometric measurements, as they are all important data. Carefully training personnel and cross-checking all data are therefore fundamental to a large-scale, long-term program such as this.

Endocrinology

The molecular sexing technique has been validated against dissected murrelets and found to be reliable in determining sex in this species (Vanderkist et al. 1999). This technique has allowed the discovery of a 2:1, male-biased sex ratio in murrelets captured by mist-netting at Desolation Sound. This bias is not found in birds captured by night-lighting at either Desolation Sound or Mussel Inlet. This result is speculated to be due to sex-specific differences in behaviour of murrelets in Theodosia Inlet during the breeding season, with males undertaking more chick provisioning than females, or using different flight paths to and from the forest (Vanderkist et al. 1999). Non- or failed breeding males may be prospecting for territories for the subsequent breeding season. Identifying these sources of bias is essential to understanding the population dynamics of the species.

The endocrinology studies by Brett Vanderkist have found that the egg yolk precursors vitellogenin and very-low density lipoprotein (VLDL) are useful indicators of egg production in marbled murrelets, with more robust results obtained when both precursors are assessed (Vanderkist 1999). This then is a reliable tool for detection of egg-producing marbled murrelets, providing data about their breeding phenology which are currently unavailable.

Habitat Usage

Colour-marking marbled murrelets using wing tags and nasal discs, and attaching radio transmitters allowed a 90% success rate in re-sighting individuals, with those with radio transmitters having the highest success rate (L. Lougheed et al. 1998*b*). A study of the effect of different methods of individually marking marbled murrelets and attaching radio transmitters revealed that nasal discs had no measurable impact on birds. Wing tags and radio transmitters have a small, short-term impact on diving duration, but the short duration of attachment of these markers would minimize any possible impacts on survival and breeding success in this species. Using wing tags and radio transmitters, which collects valuable data for management of this species, and avoids high impact activities such as tree climbing, must be balanced against any short-term effects on marbled murrelets (C. Lougheed et al. submitted).

Few marbled murrelets will be marked with nasal discs, and have radio transmitters attached in future studies. The duration of attachment of radio transmitters will be minimized. Turkey feathers will not be used in future studies as they are less effective than the previous techniques.

At-sea counts and radiotracking have found that murrelets prefer water with low sea surface temperature and high salinity early in the breeding season, but higher sea surface temperature and low salinity as the season progresses. Murrelets were also found in waters closer to old-growth forest later in the breeding season. Relatively high numbers of murrelets were observed holding fish in Theodosia Inlet compared with surrounding inlets (C. Lougheed unpubl. data), suggesting that this inlet may be an important site for foraging or stopping en route to nests. Surveys have also revealed that some murrelets remain in the Desolation Sound area to molt, and possibly remain all year.

While radiotracking has effectively determined use of the marine environment by murrelets, it was not until 1998 that it proved to be reliable for locating nest sites. Although 1 nest site was located using this technique in 1994, a larger sample size was not obtained until 1998. During 1998, radiotracking allowed the location of 23 nest sites north and east of Desolation Sound. These nests are important because they are not biased by observer expectations, hence the habitat data of these can be compared with those found from systematic surveys. These nest sites are currently being described and related to breeding success, but preliminary analyses reveal that they are in a wider range of tree species, at higher elevations, and in more inaccessible locations than those in the Bunster Range.

Nest Site and Habitat Descriptions

Sixty-two nests sites have now been found in the Bunster Range using ground searches. Nests are located primarily in high elevation stands, but this may reflect the remaining stands of old-growth forest (Drever et al. 1998). At the element scale, marbled murrelets select trees with larger diameters, higher numbers of potential nesting platforms, and higher proportions of mossy and protected platforms than other trees in the surrounding 25 m (Drever et al. 1998). Nest patches have higher densities of large diameter trees and higher numbers of platforms per tree than do random patches. There are also significantly lower densities of smaller diameter trees and total densities than random plots, suggesting that murrelets at this site may be selecting more open areas for nesting (Drever et al. 1998). Information on murrelet use of habitat in relation to elevation, biogeoclimatic classification, and tree species will be used to identify areas with high potential for intensive inventory and to ensure that habitat reserves for this species are accurately identified.

Ground surveys have found that marbled murrelet activity in the forest interior is low in May, but increases in June and July. More activity is found at the forest edge in May and early June, but activity levels are similar at all sites in July. These studies have also confirmed the re-use of nest trees by marbled murrelets.

Pilot Studies

Few marbled murrelets were captured at the Queen Charlotte Islands and this site has been abandoned. Mist-netting did not effectively catch murrelets at Mussel Inlet, but night-lighting did. Assessment of the gender of birds at Mussel Inlet revealed a 1:1 sex ratio and morphometric data allowed comparisons with birds from Desolation Sound.

Future Research and Protocols

This long-term, multidisciplinary program has seen the development of several techniques to capture and study marbled murrelets. Now that murrelets can be caught throughout the year, reliably sexed, breeding status of females determined, and nest sites located independently of ground searches, progress can be made toward understanding new aspects of the ecology of this cryptic species. These data are essential for the effective management of marbled murrelets.

Many analyses are still needed on data collected on marbled murrelets during this program. These analyses are in progress, and include the completion of Brett Vanderkist's endocrinology work for his master's degree, and marine habitat use by murrelets conducted by Cecilia Lougheed for her master's degree. Both are expected to defend in early 1999.

The capture, banding, and measurement of marbled murrelets will continue at Desolation Sound to derive better estimates of longevity, survival, site fidelity, and age at first breeding. One of the primary objectives of continuing studies at Desolation Sound is to determine the demography and improve population estimates of the species, as demographic work is essential for conservation of a species like the marbled murrelet (DeSanto and Nelson 1995, Divorky and Horton 1995), and reliable data for such analyses can only be obtained from long-term studies. Future studies of marbled murrelets will incorporate extensive training of all observers, with data continually cross-checked. The collection of samples of blood from all birds will continue as this is the most reliable way of determining the sex of murrelets, and the breeding status of females. Radiotracking has proved to be reliable in locating nests and evaluating uses of the marine environment; this aspect of the program will be expanded. Forest searches and assessment of nesting habitat will also continue, as both these methods and radiotracking improve our understanding of the habitat requirements of marbled murrelets, assisting management strategies for the species.

ACKNOWLEDGEMENTS

The program at Desolation Sound is collaborative and could not function without the involvement of numerous people. In particular, I thank F. Cooke, L. Lougheed, and the numerous field assistants who have worked on this program. I also thank Forest Renewal BC; NSERC Chair of Wildlife Ecology, Simon Fraser University; Canadian Wildlife Service; MacMillan Bloedel Ltd.; TimberWest Ltd.; International Forest Products Ltd.; Western Forest Products Ltd.; and Pacific Forest Products Ltd. for their generous financial support. Work was conducted under permits BC SCI 97/012A and 101201 AQ.

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Inland Old-Growth Rain Forests: Safe Havens for Rare Lichens?

Trevor Goward[1]
Herbarium, Department of Botany, University of British Columbia, Vancouver, BC, V6G 2B1, Canada

André Arsenault
British Columbia Ministry of Forests, Kamloops Forest Region 515 Columbia Street, Kamloops, BC, V2C 2T7, Canada

ABSTRACT

Lichens in which a cyanobacterial partner occurs can be referred to as "cyanolichens." Such species are potentially important contributors to the nitrogen budgets of some conifer forest ecosystems. In the intermontane forests of British Columbia, 31 epiphytic (tree-dwelling) cyanolichens are known to colonize conifers, including 12 species that can be considered rare or infrequent in the province as a whole. In this paper we present a simple key for predicting stand-level epiphytic cyanolichen diversity on conifers. The key is based on several readily mappable environmental factors and is useful at an operational scale. Maximum cyanolichen diversity is shown to occur in lowland old-growth rain forests established over nutrient-rich soils and subject to a rainfall pH above about 5.0. Such stands are generally restricted to the base of hill slopes in the wettest subzones of the Interior Cedar–Hemlock zone, where they not only support one of British Columbia's richest assemblages of rare cyanolichens, but also themselves represent one of the province's rarest and most endangered forest ecosystems. Further work is urgently needed.

Key words: biodiversity, cyanolichens, epiphytes, inland rain forests, lichens.

Lichens can be described, with only slight exaggeration, as the 20% of fungi that have "discovered agriculture." Whereas most fungi satisfy their carbohydrate requirements largely through the absorption of nutrients from the same organic matter within which they pass their lives (decaying wood, humus, etc.), lichenized fungi cultivate their foodstuff internally; that is, among the tightly woven weft of fungal threads which comprise them. In effect, lichens are "living fungal greenhouses" in which tiny photosynthesizing cells, collectively referred to as photobionts, are simultaneously cultivated and assimilated. To date, approximately 100 algal and cyanobacterial species in 40 genera are known to act as lichen photobionts. In 90% of lichens, the photobiont is an alga. In other species, however, the lichen fungi associate exclusively or in part with cyanobacteria; for convenience, such species can be referred to as "cyanolichens."

Cyanolichens are nitrogen-fixers; they both absorb nitrogen from the atmosphere and, ultimately, release it into their immediate environment. This ability has stimulated considerable research on the potential of cyanolichens to contribute significantly to the nitrogen budgets of some ecosystems. Though conclusive experimental evidence is lacking (Nash 1996), there is growing consensus that at least epiphytic (tree-dwelling) cyanolichens probably do make a significant contribution under some circumstances. Indeed, such species are now generally assumed to be keystone organisms in some conifer forests in which nitrogen is limiting (Rhoades 1995).

Several lichen species considered to be rare or infrequent in British Columbia take the form of epiphytic cyanolichens. If only for this reason, it would be useful to develop a methodology for predicting their general levels of occurrence in different forest ecosystems. If founded on readily mappable environmental factors, such a methodology could provide a useful tool for resource managers charged with maintaining forest biodiversity at historic levels. In this paper, we attempt to contribute to a basic understanding of cyanolichen distributional ecology in intermontane British Columbia. Our specific objectives are: 1) to summarize the most important environmental factors promoting cyanolichen diversity in inland forests; and 2) to propose a simple method by which cyanolichen diversity can be reliably predicted at an operational scale, based on readily available information.

STUDY AREA

The study area encompasses all of intermontane British Columbia, comprising roughly 75% of the surface area of the province (Fig. 1). This is a highly heterogeneous region, in

1 Present address: Enlichened Consulting, Edgewood Blue P.O. Box 131, Clearwater, BC V0E 1N0 tgoward@mail.wellgray.net

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

which 3 broad "ecodivisions" (Demarchi et al. 1990) can be recognized, as well as 12 climatic regions (Marsh 1988) and 11 of British Columbia's 14 biogeoclimatic zones (Meidinger and Pojar 1991). Included here are some of the wettest, driest, warmest, and coldest portions of southern inland Canada (Phillips 1990). We have restricted our study to regions having an "intermediate" continentality (Tuhkanen 1984); that is, areas with continentality ratings (Conrad 1946) between about 30 and 45. The Nass and Skeena valleys (continentality <30) are therefore not considered; nor are regions lying north and east of the Rocky Mountains (continentality >45).

METHODS

The distributional patterns reported here are based largely on field studies conducted by the senior author over 20 years and summarized, in part, in Goward (1994, 1995, 1996, 1999), Goward and Ahti (1992, 1997), Goward and Arsenault (1997), Goward et al. (1994), as well as in several unpublished studies undertaken in the Kamloops, Nelson, Prince George, Prince Rupert, and Vancouver forest regions. In addition, more than 40,000 herbarium specimens housed at the Canadian Museum of Nature in Ottawa, the University of British Columbia in Vancouver, and the University of Victoria in Victoria have been examined and mapped. More recently, the authors have assembled quantitative data on 40 stands supporting epiphytic cyanolichens. For a summary of our field methods, see Goward and Arsenault (1997). The taxonomy and nomenclature adopted in this paper follow Esslinger and Egan (1995).

RESULTS AND DISCUSSION

Epiphytic cyanolichens are by no means randomly, or evenly, distributed in intermontane British Columbia. Rather, their occurrence appears to be closely associated with a finite number of environmental factors. Based on our field

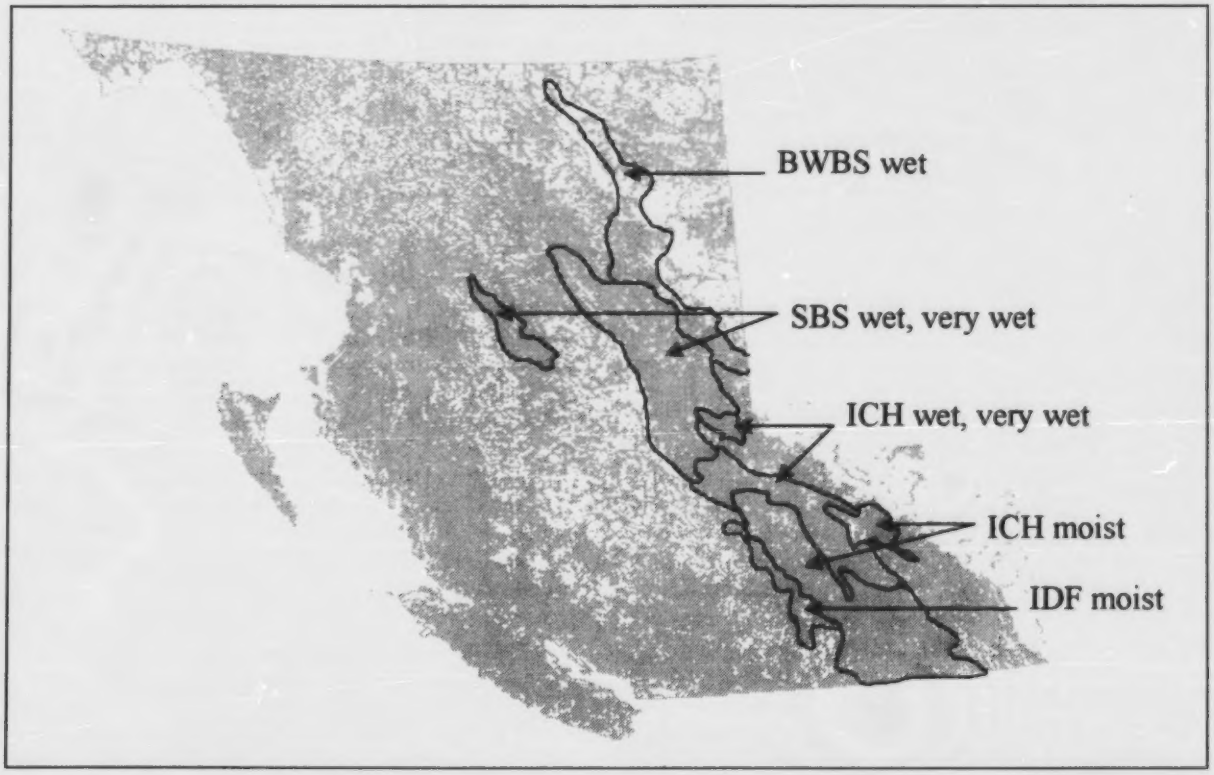

Figure 1. Occurrence of epiphytic cyanolichens on conifer branches in intermontane British Columbia. BWBS = Boreal White and Black Spruce zone (wet subzone only); IDF = Interior Douglas-fir zone (moist and wet subzones only); SBS = Sub-boreal Spruce zone (wet and very wet subzones only); ICH "inland" = Interior Cedar–Hemlock zone (moist subzones, and wet and very wet subzones only); and ICH "coast" (occurs outside study area: included for comparative purposes only).

work in the study area, as well as on research conducted in other parts of the world, at least 7 such factors can be recognized. Four of these can be considered "primary" because they appear to be prerequisite to the colonization of conifers by cyanolichens. They are: 1) air quality; 2) elevation; 3) biogeoclimatic zonation; and 4) nutrient enrichment.

Primary Cyanolichen Factors

Air Quality

Epiphytic cyanolichens are generally restricted to the bark of trees in which surface pH registers between about 5.0 and 6.0 (Gauslaa 1985). When acid rain or other factors depress bark pH below about 5.0, these species go into decline (Farmer et al. 1991, 1992). In parts of Europe in which acid rain is now prevalent, striking declines in epiphytic cyanolichen diversity have been reported in forests otherwise unaffected by human activity (e.g., Gilbert 1986, Hallingbäck and Olsson 1987, Tnsberg et al. 1996). To date, most authors have emphasized the adverse effects of acid rain on the epiphytes of deciduous trees. Yet substrate acidification is probably even more devastating to the cyanolichen floras of conifers, because of the low buffering capacity of the bark (Barkman 1958, Gauslaa and Holien 1998). Apparently reflecting this, cyanolichens have been reported from conifers only in portions of the world unaffected by acid rain, including intermontane British Columbia, where rainfall acidity is >5.0 (B.C. Ministry of Environment 1990). Elsewhere, if epiphytic cyanolichens occur at all, they are restricted primarily to the trunks and branches of deciduous trees having strongly buffered bark (Rose 1988).

Elevation

Few studies have been undertaken on lichen distribution in upper elevation forests of inland British Columbia. What little information does exist is contained incidentally in the floristic studies of Goward and Ahti (1992) and Goward and Arsenault (1997); these studies suggest that epiphytic cyanolichens are essentially absent from all upper forested zones, including the Engelmann Spruce–Subalpine Fir, the Montane Spruce, the Sub-boreal Pine–Spruce, and the Spruce–Willow–Birch zones (Meidinger and Pojar 1991). Even in sheltered, valley-bottom forests, cyanolichens tend to disappear above about 1,100 m in southern regions, and above 500–900 m farther north. Sillett and Neitlich (1996) and Neitlich and McCune (1997) report a similar phenomenon in coastal regions of the American northwest. From these observations, we conclude that epiphytic cyanolichens are optimally represented at lower elevations.

Nutrient Enrichment

There is mounting evidence that cyanolichens colonize conifer branches most abundantly in stands growing over nutrient-rich soils—presumably due to enhanced bark pH (Gauslaa 1985, Gauslaa and Holien 1998). While the precise mechanism(s) by which these 2 phenomena are linked has/have not been elucidated, Gauslaa and Holien (1998) suggest that some conifers may be capable of absorbing calcium at concentrations sufficient to promote cyanolichen growth; this hypothesis, however, requires testing. Goward and Arsenault (in press) argue that a leachate or "drip zone" effect may be involved in some cases (see also Arsenault and Goward 2000). According to this hypothesis, aspen and cottonwood (*Populus* spp.) often act as "nutrient pumps." In nutrient-rich sites, at least, the roots of these trees are capable of absorbing calcium, nitrogen, and magnesium in considerable quantities (Krajina and Klinka 1982, Peterson et al. 1996, Holien 1997). Carried into the upper canopy, these nutrients are eventually released (as leaf litter and canopy leachates) to other ecosystem components, including the branches of understory conifers. In this way the pH of conifer branches might be enhanced, and the establishment of epiphytic cyanolichens promoted. Another localized form of enrichment results from aerosols carried in the spray from waterfalls.

Biogeoclimatic Zonation

Most epiphytic cyanolichens can be characterized as hygrophytic; that is, they require prolonged wetting at rather frequent intervals. Whereas alga-containing lichens can achieve positive net gas exchange when exposed to water vapour alone, a corresponding effect in cyanolichens requires direct exposure to liquid water (Lange et al. 1986, 1993). Consistent with these observations, Figure 1 reveals that epiphytic cyanolichens are primarily restricted to the humid, windward slopes of the major mountain ranges. Similarly, Table 1 indicates a 3- to 6-fold increase in cyanolichen diversity in the wettest inland subzones of the Interior Cedar–Hemlock (ICH) zone, as compared with adjacent but slightly drier zones and subzones. This increase, moreover, continues into more "oceanic" subzones of the ICH zone located outside the study area. Thus, regions subject to humid, oceanic climates appear to be optimal for epiphytic cyanolichens.

Secondary Cyanolichen Factors

To the 4 primary cyanolichen factors discussed above, we can add 3 secondary factors: forest age; soil moisture regime; and stand spacing. While not prerequisite to cyanolichen colonization on conifer branches, these factors do promote the establishment of nodes of high cyanolichen diversity. Several rare species are restricted to such nodes.

Forest Age

Old-growth forests are important to many epiphytic cyanolichens. Not only are cyanolichens more abundant in old-growth forests than in younger forest stands (Lesica et al. 1991, McCune 1993, Esseen et al. 1996), they are also often

Table 1. Epiphytic cyanolichens of conifer branches in intermontane British Columbia. Occurrences accompanied by an asterisk are old-growth-associated in the subzones indicated. Occurrences denoted by parentheses are restricted to the spray zones of waterfalls. Zonal abbreviations as in Figure 1.

	Zone and subzone(s)					
Cyanolichens [a,b]	BWBS Wet	IDF Moist Wet	SBS Wet Very Wet	ICH Moist	ICH Wet Very Wet	ICH "Oceanic"[c]
Collema auriforme					+	
Collema furfuraceum				+	+	+
Collema subflaccidum				(+)	(+)	
Fuscopannaria ahlneri						+*
Fuscopannaria leucostictoides					+*	+*
Fuscopannaria mediterranea				(+)	(+)	+*
Leptogium burnetiae				(+)	+	+
Leptogium pseudofurfuraceum		+				
Leptogium saturninum	+	+	+	+	+	+
Lichinodium canadense					+*	+*
Lobaria hallii				(+)	+*	+
Lobaria linita					+*	+*
Lobaria oregana					(+)	+*
Lobaria pulmonaria		+	+	+	+	+
Lobaria retigera					+*	+*
Lobaria scrobiculata				(+)	+	+
Lobaria silvae-veteris						+*
Nephroma bellum	+			+	+	+
Nephroma helveticum			+	+	+	+
Nephroma isidiosum				(+)	+*	+*
Nephroma occultum					+*	+*
Nephroma parile			+	+	+	+
Nephroma resupinatum	+		+	+	+	+
Parmeliella parvula				(+)	(+)	+
Parmeliella triptophylla				(+)	+	+
Peltigera collina				(+)	+	+
Polychidium contortum						+*
Polychidium dendriscum					+*	+*
Pseudocyphellaria anomala				+	+	+
Pseudocyphellaria crocata					(+)	
Spilonemella americana [d]					(+)	+*
Sticta fuliginosa			+	+	+	+
Sticta limbata				(+)	+*	+*
Sticta oroborealis					+*	+*
Sticta wrightii					+*	+*
TOTAL SPECIES	3	3	6	8(11)	25(6)	30

[a] Species in bold are considered to be rare or infrequent in intermontane regions.

[b] Underlined species are rare or infrequent in British Columbia as a whole.

[c] Located in the Skeena and Nass river valleys, and transitional to the Coastal Western Hemlock zone; included for comparative purposes only.

[d] See "Unknown 1" in Goward (1999).

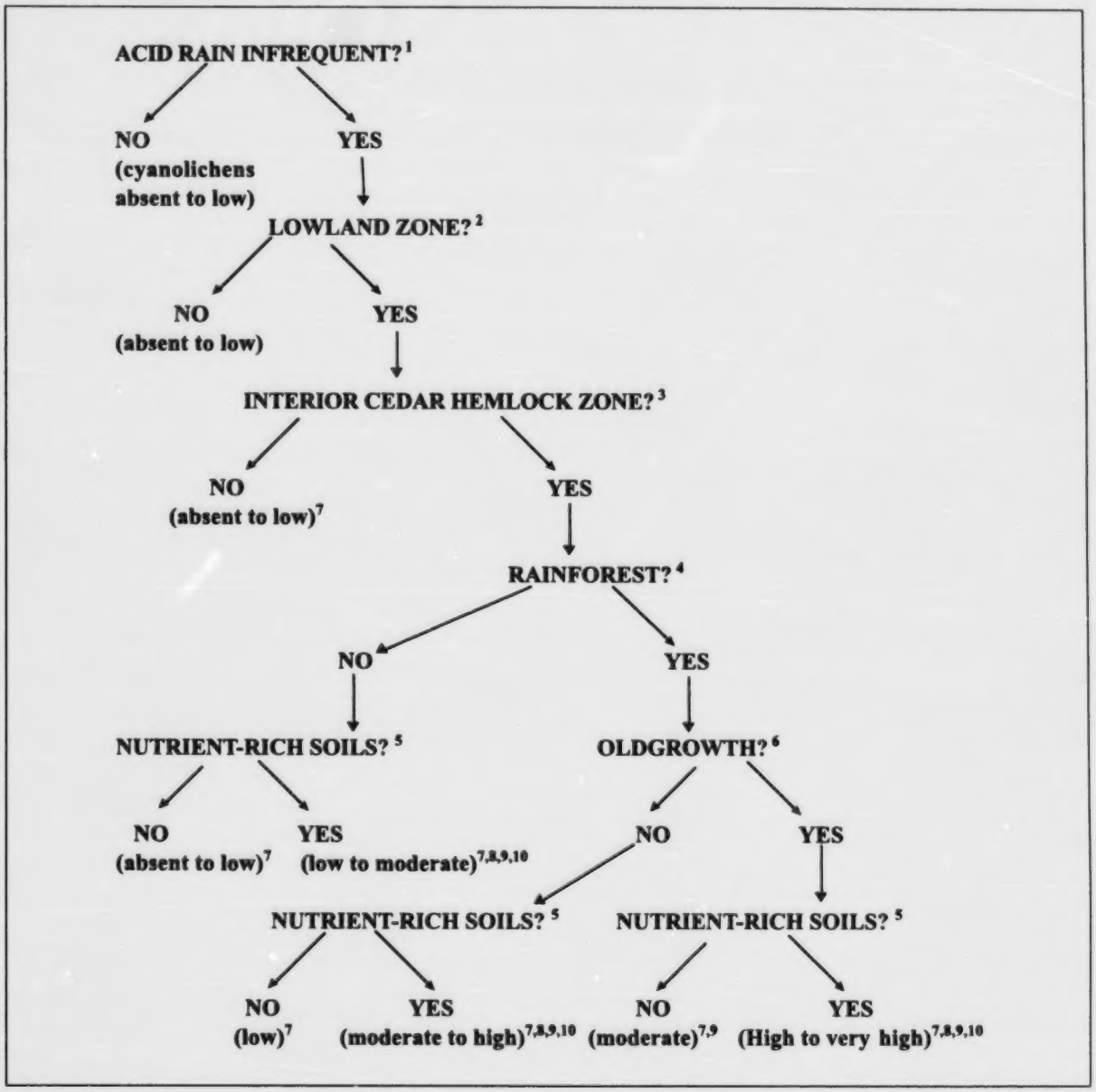

Figure 2. Provisional key to epiphytic cyanolichen diversity on conifer branches in intermontane British Columbia. Notes: [1]Rainfall pH averages >5.0; [2]Bunchgrass zone, Ponderosa Pine zone, Interior Douglas-fir zone, Interior Cedar–Hemlock zone, Subboreal Spruce zone, Boreal Black and White Spruce zone, and Spruce–Willow–Birch zone (Meidinger and Pojar 1991); [3]Western redcedar (*Thuja plicata*) and/or western hemlock (*Tsuga heterophylla*) present in mature stands on mesic sites; [4]Oak fern (*Gymnocarpium dryopteris*) and/or devil's club (*Oplopanax horridus*) often present in mature stands on mesic sites; [5]Aspen or cottonwood (*Populus* spp.) present, especially in the "toe" position at the base of slopes; [6]Snags and trees in all age classes present; *Populus* sparse or absent; [7]Occurrence within the spray zones of waterfalls can enhance cyanolichen diversity, assuming nutrient-rich aerosols; [8]Occurrence within the "drip zone" of *Populus* can enhance cyanolichen diversity, assuming nutrient-rich soils; [9]Hygric sites often support higher cyanolichen diversity than mesic or xeric sites; [10]Open stands support higher cyanolichen diversity than closed stands.

more diverse (Goward 1994, Goward and Pojar 1998). Goward (1994) has hypothesized that old old-growth (or "antique") forests act as "ports of entry" for rare epiphytic lichens colonizing from great distances. At least 16 cyanolichen species are essentially restricted to old-growth forests in the study area (Table 1); in the absence of old-growth, many of these species would not occur in this portion of British Columbia (Goward 1995). In coastal regions, too, many cyanolichens are restricted to older forests (Sillett and Neitlich 1996).

Soil Moisture Regime

In inland regions, mesic to hygric sites are more favourable to cyanolichen colonization than are submesic to xeric sites.

Stand Spacing

Open conifer stands are more conducive to cyanolichen development than are dense, shady stands (Neitlich and McCune 1997).

KEY TO CYANOLICHEN DIVERSITY

Our field work suggests that epiphytic cyanolichen diversity on conifers is roughly proportionate to the number of primary factors overlapping within a given forest stand. Maximum diversity is thus usually restricted to stands in which all 4 primary factors co-occur. We stress, however, that cyanolichen richness in any given stand can be further enhanced by the co-occurrence of 1 or more secondary factors. The seven cyanolichen factors discussed above can be arranged in a rough hierarchy of increasing influence, as follows: stand spacing < soil moisture regime < proximity to *Populus*/proximity to waterfalls < forest age < soil nutrient status < biogeoclimatic zonation < elevation < air quality. Guided by this schema, we have prepared a provisional "key" to epiphytic cyanolichen diversity on conifer branches in intermontane British Columbia (Fig. 2). Our key provides a simple method for the rapid assessment of cyanolichen richness. It is intended for use at an operational scale (e.g., 1:20,000–1:50,000) and should prove helpful to ecologists, forestry planners, resource managers, and others who may wish to adopt cyanolichens as surrogates for biodiversity, environmental continuity, or other ecological values.

CONCLUSIONS

The conifer forests of intermontane British Columbia support at least 31 cyanolichen species—one of the richest epiphytic cyanolichen assemblages in the world. Most of these cyanolichens are restricted in inland regions to "rain forests" located in the wet and very wet subzones of the ICH zone (Table 1). Within these subzones, moreover, the distributions of 16 species are further confined to old-growth ecosystems (Table 1). Virtually all of these species can be considered rare or infrequent in intermontane regions, while at least 12 are rare or infrequent in British Columbia as a whole: *Collema auriforme*, *Fuscopannaria leucostictoides*, *F. mediterranea*, *Lichinodium canadense*, *Lobaria retigera*, *Nephroma isidiosum*, *N. occultum*, *Parmeliella parvula*, *Polychidium dendriscum*, *Spilonemella americana*, *Sticta oroborealis*, and *S. wrightii* (Goward 1996, Goward 1999).

We conclude, then, that maximum cyanolichen diversity in inland British Columbia occurs over nutrient-rich soils in lowland old-growth rain forests subject to rainfall pH >5.0. Such stands are characteristically situated in nutrient-receiving sites, especially the "toe" position at the base of slopes. Here wildfire occurs infrequently, and stands as old as 800 years have been recorded. Reflecting their topographic position, however, such stands are highly vulnerable to road construction, timber extraction, and flooding by hydroelectric dams. Many have already disappeared and most of those that remain are now highly fragmented. Once safe havens for rare lichens, British Columbia's richest inland old-growth rain forests currently number among the most endangered of the province's endangered ecosystems. We hope the simple method advocated in this paper will assist resource managers in locating and assessing the last remaining stands, whether for designation as protected areas, old-growth management areas, or special management zones.

ACKNOWLEDGEMENTS

We are grateful to A. Roberts for data on the occurrence of epiphytic cyanolichens in the Interior Douglas-fir zone, and to E. Hamilton and L. Kowalsky for comments on an earlier draft of this paper. Funding for this project was provided by Forest Renewal British Columbia.

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The Drip Zone Effect: New Insights Into the Distribution of Rare Lichens

André Arsenault
British Columbia Ministry of Forests, Kamloops Forest Region
515 Columbia Street, Kamloops, BC, V2C 2T7, Canada
Andre.Arsenault@gems8.gov.bc.ca

Trevor Goward
Enlichened Consulting, Edgewood Blue
Box 131, Clearwater, BC, V0E 1N0, Canada

Key words: cyanolichen, lichen.

EXPANDED ABSTRACT

Nearly half of British Columbia's rare tree-dwelling macrolichens have a cyanobacterium as photobiont. Such species can be referred to as epiphytic cyanolichens. As a group, epiphytic cyanolichens have a requirement for nutrient-rich substrates with a pH above about 5.0 (Gauslaa 1985); they tend to be absent from acidic substrates, including the bark of conifers. Viewed from this perspective, the copious presence of cyanolichens over the bark of pines, spruces, hemlocks, and other members of the Pinaceae in some portions of northwestern North America would seem anomalous. We propose that this phenomenon must reflect nutrient enrichment from sources extraneous to the trees themselves. Enrichment may derive, for example, from airborne dust, aerosols associated with the spray zones of waterfalls, or from nutrients present in the soil.

Our work in various subzones of the Interior Cedar–Hemlock biogeoclimatic zone suggests that nutrient transfer from the soil into the canopy is mediated at least in part by the roots of cottonwood trees and trembling aspen. We hypothesize that such trees can effectively function as "nutrient pumps." The nutrients transported into the canopy by them are later released into the surrounding forest in what may be called a "drip zone effect" (Fig. 1). Though cottonwood and aspen do not themselves support diverse assemblages of epiphytic cyanolichens, they do permit the development of impressive nodes of cyanolichen diversity on conifers growing adjacent to them. By their presence, they indirectly provide habitat for many of British Columbia's rarer epiphytic species. If it is true that nitrogen-fixing cyanolichens play a significant role in the nitrogen budgets of some forest ecosystems (Pike 1978), then aspen and cottonwood should be considered "keystone" organisms.

Viewed globally, the ability of cyanolichens to colonize conifers has probably declined during the past 2 centuries. In most parts of Europe, for example, acid rain has been on the rise for more than 100 years, and conifers no longer support epiphytic cyanolichens (Gauslaa and Holien 1998). Instead, epiphytic cyanolichens are restricted to the bark of deciduous trees with a high buffering capacity (Rose 1988).

The situation is very different in most portions of British Columbia. Here, air quality remains high, and cyanolichens continue to thrive in at least some conifer woodlands (Goward and Arsenault 2000). We tentatively conclude that the study of conifer-cyanolichen relationships in western North America may yield important insights into pre-industrial epiphytic ecology over large portions of mid-latitude North America and Europe. Moreover, the intricate relationship between *Populus* and cyanolichen/nitrogen distribution may be expected to have significant implications for silvicultural practices in British Columbia. These ideas will be developed more fully elsewhere.

ACKNOWLEDGEMENTS

We extend thanks to D. Burgess and G. Yearsley for assistance in the field.

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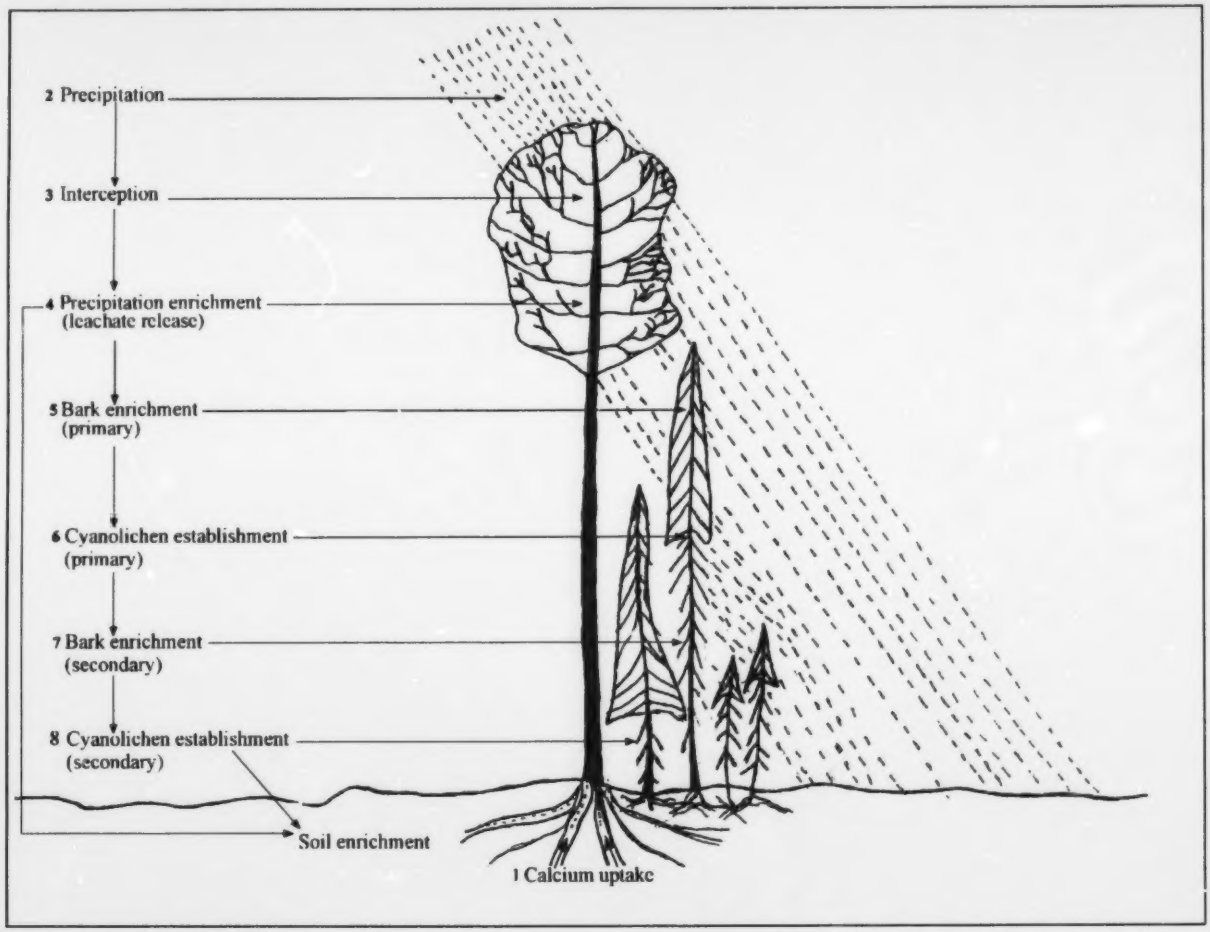

Figure 1. The drip zone hypothesis.

Floristics and Distribution Patterns of Lichens and Bryophytes in Microbiotic Crusts of British Columbia's Ponderosa Pine Forests

Patrick Williston
Department of Botany, University of British Columbia
3529 - 6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada
pwilliston@yahoo.com

ABSTRACT

Microbiotic crusts are an important though often overlooked component of the biodiversity and ecology of arid and semiarid ecosystems. This study lists 51 terricolous lichens and 39 terricolous bryophytes of microbiotic crusts in British Columbia's semiarid ponderosa pine (*Pinus ponderosa*) forests. The list is the first to document the terricolous lichens of these forests, and adds 14 species to previous reports of the bryophytes. This study also describes successional patterns, following grazing disturbance, of invasive nonnative vascular plants, 3 native grasses, and 3 microbiotic crust species. An understanding of successional patterns is essential for managing the biodiversity of British Columbia's ponderosa pine forests, one of the most restricted ecosystems in the province.

Key words: bryophyte, lichen, microbiotic crust, *Pinus ponderosa*, ponderosa pine.

BACKGROUND

Microbiotic crusts, also called cryptogamic, cryptobiotic, microphytic, biologic, and microfloral crusts, are an important component of the biodiversity and ecology of arid and semiarid ecosystems. Unfortunately, they are also often overlooked in ecological studies and vegetation inventories. Microbiotic crusts are assemblages of bryophytes, lichens, algae, fungi, bacteria, and cyanobacteria that integrate with the soil surface of arid and semiarid ecosystems. While earlier studies have reported the common bryophytes in ponderosa pine (*Pinus ponderosa*) forests of British Columbia, little work has been conducted to date on the lichens, algae, fungi, bacteria, and cyanobacteria occurring in this ecosystem (Schofield 1988). This study adds to existing lists of the bryophytes of ponderosa pine forests and is the first to report the common terrestrial lichens in these ecosystems.

Inquiry into the ecological functions of microbiotic crusts derives from studies conducted in the United States and Australia (West 1990; Eldridge 1993, 1996; Harper and Pendelton 1993; Belnap 1994; Eldridge and Greene 1994; Eldridge and Tozer 1996; Eldridge et al. 1997). In these regions, microbiotic crusts have been shown to influence nitrogen balance, water infiltration and retention, and water and wind erosion in grassland ecosystems (West 1990, Eldridge 1993, Harper and Pendleton 1993). Recent research in the grasslands of the Okanagan Valley of British Columbia determined that microbiotic crusts have a significant effect on water retention after rain events (Atwood 1998). Few studies have examined successional patterns in crust communities (Looman 1964). The research presented here is the first to describe patterns of succession in British Columbia's ponderosa pine forests.

Microbiotic Crusts

Despite the rich diversity of bryophytes and lichens in the semiarid interior of British Columbia, our understanding of its flora and ecology is rudimentary. McIntosh (1986, 1997) conducted the most complete survey of the bryophytes of British Columbia's grasslands in a thesis that included a key to grassland bryophytes and reported several species new to the province. While his study did not include ponderosa pine ecosystems, several of the species overlap from the grasslands to the ponderosa pine forests. Schofield (1988) reported species belonging to ponderosa pine forests in a general survey of the biogeoclimatic zones of British Columbia. The present study adds 14 species to the existing list and provides insights into the ecology of 1 dominant bryophyte.

Lichens have been the focus of intensive investigation in 4 regions of the province: the coastal Douglas-fir zone (Noble 1982), the Queen Charlotte Islands (Brodo 1995), and wet coastal and interior ecosystems (Goward and Thor 1992, Goward et al. 1994, 1996). This research lists the terrestrial lichens from 19 ponderosa pine study sites and discusses successional patterns of 2 common lichens, *Cladonia pyxidata* and *C. chlorophaea*. Efforts to describe crust communities have been more thorough in the Great Basin region of the United States (St. Clair et al. 1993). While several species from the Great Basin are shared with the British Columbia flora, genera such as *Cladonia* appear to increase

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in diversity and cover with latitude, making British Columbia's flora unique (Goward and Ahti 1997).

Ponderosa Pine Forests

British Columbia's ponderosa pine forests are restricted to the lowlands and low slopes of the driest valleys in the south-central and southeastern regions of the province. The mean annual rainfall is 280–500 mm (Meidinger and Pojar 1991). These forests are characterized by hot, dry summers and cool winters with little snowfall (Meidinger and Pojar 1991). The canopy is comprised of broadly spaced ponderosa pine, with Douglas-fir (*Pseudotsuga menziesii*) appearing in draws and wetter areas. The understory vegetation is dominated by the perennial bunchgrasses *Festuca campestris* and *Elymus spicata*, and includes a diverse array of forbs and shrubs.

Historically, these forests were subject to natural periodic fires; however, grazing has since replaced fire as the primary source of disturbance, resulting in an increase in the dominance of early-successional vegetation. Turf-forming *Poa pratensis* and the annual *Bromus tectorum*, both of which are aggressive, nonnative graminoids, have replaced native perennial bunchgrasses in areas of heavy disturbance (Daubenmire 1968, 1970). Indeed, ponderosa pine forests in a natural condition are rare in British Columbia—a conservation issue worthy of concern.

METHODS

Study Area

The study was conducted along a glacial lake-bed terrace east of Kamloops, B.C. The sites were oriented along an 18-km-long east/west transect on the south side of the South Thompson River. The elevation ranged between 500 and 600 m, and slopes remained between 2 and 15%, with a north aspect. Sites represented early (E), mid (F), late (L), and natural (P) seral stages and were restricted to a zonal position in the Ponderosa Pine biogeoclimatic zone (Lloyd et al. 1990). Site history and the condition and composition of the dominant vegetation were used to determine seral stages. Five sites were sampled at each seral stage except the mid-seral stage, which was sampled at 4 sites. In total there were 19 sites.

Sampling

Fieldwork was conducted using a frame with 10 precise sampling points. The frame was positioned at random intervals along 5 50-m transects on a 40-m baseline. Each transect was sampled with 10 placements of the sampling frame to yield 100 points per transect. These points were used to determine the percent cover of vascular plants and microbiotic crust species. Crust specimens were curated as described by Rosentreter et al. (1988).

RESULTS AND DISCUSSION

Bryophyte and Lichen Survey

Table 1 lists the terricolous bryophytes and lichens observed while conducting this research, including 14 species of terrestrial bryophytes not previously reported for the ponderosa pine forests of British Columbia (Schofield 1988). This list includes those species observed in the study area and does not constitute a complete survey of the ponderosa pine forest ecosystem. Voucher specimens of these species are located at the University of British Columbia herbarium and the British Columbia Ministry of Forests, Range Branch in Victoria. Williston (1999) provides an illustrated key to the common bryophytes and lichens of the ponderosa pine forests and associated grasslands of British Columbia.

Vascular Plants

The successional transition of forested grassland ecosystems from nonnative annual grasses to native bunchgrasses in the British Columbia Interior has been previously described (McLean and Marchand 1968). This study builds upon existing knowledge by examining some of the ecological effects of these species' distributions and vegetation structure changes. Early successional vegetation is comprised of native and nonnative annual grasses with a high cover of the native perennial *Stipa comata* (Figs. 1 and 2). Over time, the cover of *S. comata* and nonnative grasses decreases and the cover of large native perennial bunchgrasses such as *Festuca campestris* and *Elymus spicata* increases (Figs. 3 and 4). The shrub *Artemesia tridentata*, a species known to be unpalatable to cattle (the primary nonnative agent of disturbance in this ecosystem), increases in cover under intensive grazing management strategies (Daubenmire 1970). *A. tridentata* provides shade in early seral sites; however, the chemistry of its litter differs from that of the bunchgrasses. This is reflected in the distribution patterns of microbiotic crust species, which suggest that some late successional species, such as *Brachythecium albicans*, are able to colonize under the cover of *A. tridentata*, while others are not (Fig. 7).

Microbiotic Crust Species

Cladonia pyxidata is 1 of the most common crust species in the ponderosa pine forests of British Columbia. This species increases in cover in the mid-seral stage and then decreases with succession (Fig. 5). *C. pyxidata* is a soil colonizer and is able to utilize soil substrata created by natural and human-caused disturbances. An increase in substratum availability in the mid-seral stage probably accounts for the greater cover of *C. pyxidata*.

Cladonia chlorophaea is most commonly found beneath the grass litter of native perennial bunchgrasses such as *Festuca campestris* and *Elymus spicata*. It is not surprising that this species increases in cover in later seral stages where bunchgrasses dominate the understory (Fig. 6).

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Table 1. A list of the terricolous bryophytes and lichens from the study area in the ponderosa pine forests of the southern interior of British Columbia.

Liverworts

Barbilophozia hatcheri (Evans) Loeske*
Cephaloziella divaricata (Sm.) Schiffn.
Lophozia excisa (Dicks.) Dum.*
Ptilidium ciliare (L.) Hampe*
Ptilidium pulcherrimum (G. Web.) Hampe*

Mosses

Aloina brevirostris (Hook & Grev.) Kindb.
Barbula convoluta Hedw.
Brachythecium albicans (Hedw.) Shimp. in B.S.G.
Brachythecium collinum (Schleich. ex C. Mull.) Shimp. in B.S.G.
Bryum argenteum Hedw.
Bryum caespiticium Hedw.
Buxbaumia aphylla Hedw.*
Ceratodon purpureus (Hedw.) Brid.
Desmatodon convolutus (Brid.) Grout
Dicranum polysetum Sw.*
Dicranum scoparium Hedw.
Didymodon vinealis (Brid.) Zander
Ditrichum flexicaule (Swaegr.) Hampe*
Encalypta rhaptocarpa Schwaegr.*
Encalypta vulgaris Hedw.*
Fissidens bryoides Hedw.*
Funaria hygrometrica Hedw.
Homalothecium aeneum (Mitt.) Lawt.
Hylocomium splendens (Hedw.) B.S.G.
Hypnum revolutum (Mitt.) Lindb.
Hypnum vaucheri Lesq.
Mnium spinulosum B.S.G.*
Phascum cuspidatum Hedw.
Pohlia nutans (Hedw.) Lindb.
Polytrichum juniperinum Hedw.
Polytrichum piliferum Hedw.
Pseudoleskeella tectorum (Brid.) Broth.
Pterygoneurum subsessile (Brid.) Jur.
Pterygoneurum ovatum (Hedw.) Dix.
Rhytidiadelphus triquetrus (Hedw.) Warnst.
Rhytidiopsis robusta (Hedw.) Broth.*
Sanionia uncinatus Loeske*
Tortula ruralis (Hedw.) Gaertn. et al.
Weissia hedwigii Crum*

Lichens

Acarospora schleircheri (Ach.) A. Massal.
Amandinea punctata (Hoffm.) Coppins & Scheid
Arthonia glebosa Tuck.
Bryonora castanea (Hepp) Poelt
Buellia geophilla (Florke ex Sommerf.) Lynge
Caloplaca jungermanniae (Vahl) Th. Fr.
Caloplaca tominii Savicz
Candelariella terrigena Rasanen
Catapyrenium squamulosum (Ach.) Bruess
Cetraria ericetorum Opi
Cladina arbuscula (Wallr.) Hale & Culb.
Cladina mitis (Sandst.) Hustich
Cladina rangiferina (L.) Nyl.
Cladonia borealis S. Stenroos
Cladonia botrytes (K. Hagen) Willd.
Cladonia cenotea (Ach.) Schraerer
Cladonia cervicornis (Ach.) Flot.
Cladonia chlorophaea (Florke ex Sommerf.) Spreng.
Cladonia cornuta (L.) Hoffm.
Cladonia ecmocyna Leighton
Cladonia fimbriata (L.) Fr.
Cladonia gracilis (L.) Willd.
Cladonia mulitformis G. Merr.
Cladonia phyllophora Hoffm.
Cladonia pocillum (Ach.) Grognot
Cladonia pyxidata (L.) Hoffm.
Cladonia rei Schraerer
Cladonia subulata (L.) F. H. Wigg.
Cladonia sulphurina (Michaux) Fr.
Cladonia symphycarpa (Florke) Fr.
Collema tenax var. *crustaceum* (Kremp.) Degel.
Diploschistes muscorum (Scoop) R. Scant
Endocarpon pusillum Hedw.
Leptogium sp.
Nephroma parile (Ach.) Ach.
Ochrolechia upsaliensis (L.) A. Massal.
Peltigera canina (L.) Willd.
Peltigera didactyla (With.) Laundon
Peltigera didactyla var. *extenuata* (Nyl.) Goffinet & Hastings
Peltigera lepidophora (Vainio) Bitter
Peltigera malacea (Ach.) Funck
Peltigera ponojensis Gyel.
Peltigera rufescens (Weis) Humb.
Physconia muscigena (Ach.) Poelt
Placynthiella uliginosa (Schrader) Coppins & P. James
Polychidium muscicola (Sw.) Gray
Psora cerebriformis W. A. Weber
Psora globifera (Ach.) A. Massal.
Psora montana Timdal
Stereocaulon sp.
Xanthoria wyomingica (Gyel.) Hale

* Denotes species added to reported lists of bryophytes of British Columbia's ponderosa pine forests.

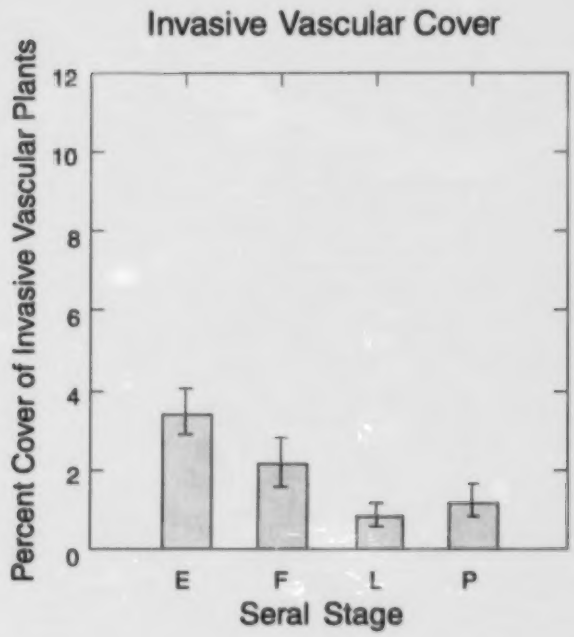

Figure 1. Percent cover of invasive, nonnative vascular plants over 4 successional stages: early (E), mid (F), late (L), and natural (P). Cover is the mean per transect.

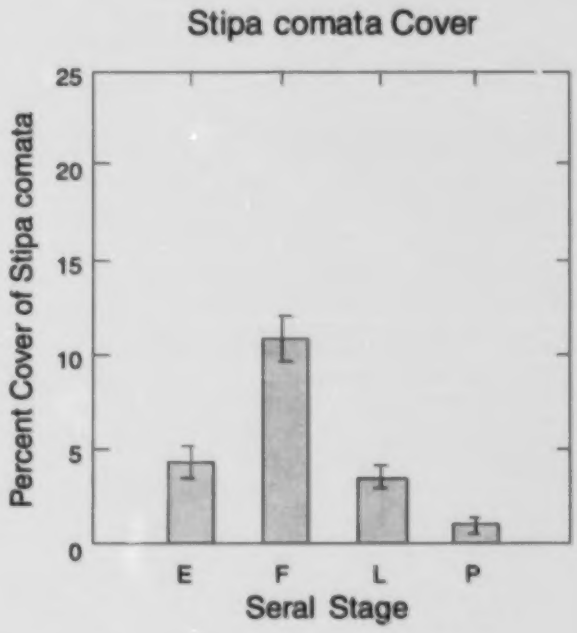

Figure 2. Percent cover of *Stipa comata* over 4 successional stages, as described in Figure 1.

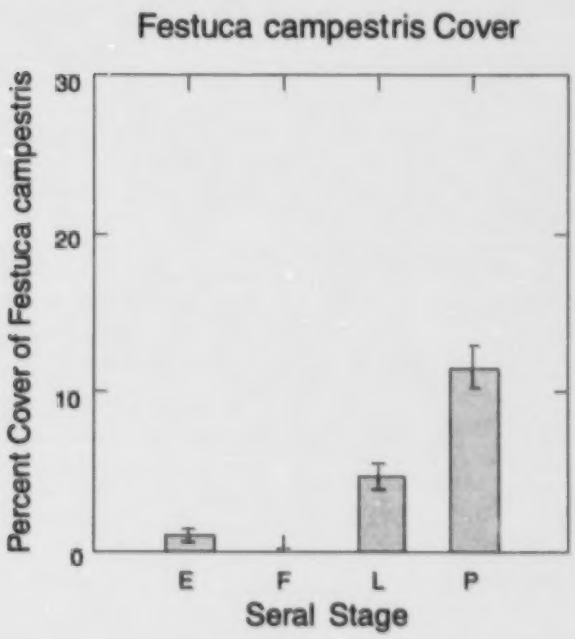

Figure 3. Percent cover of *Festuca campestris* over 4 successional stages, as described in Figure 1.

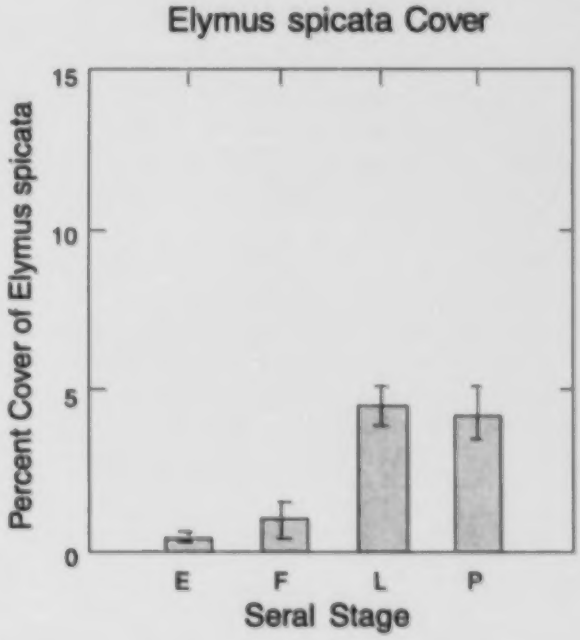

Figure 4. Percent Cover of *Elymus spicata* over 4 successional stages, as described in Figure 1.

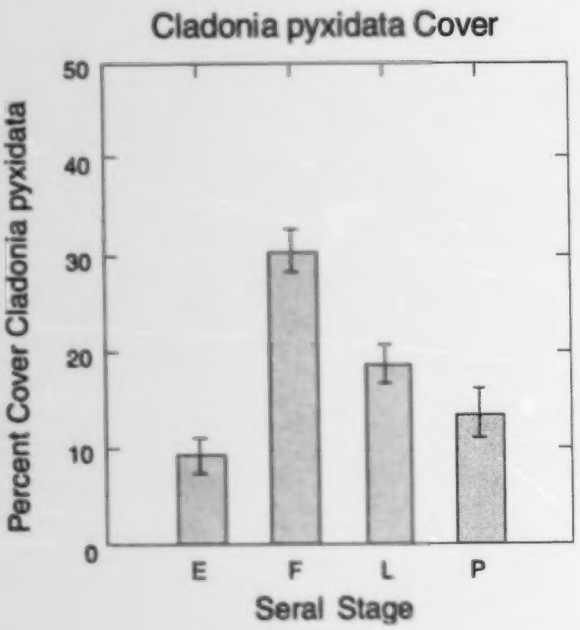

Figure 5. Percent cover of *Cladonia pyxidata* over 4 successional stages, as described in Figure 1.

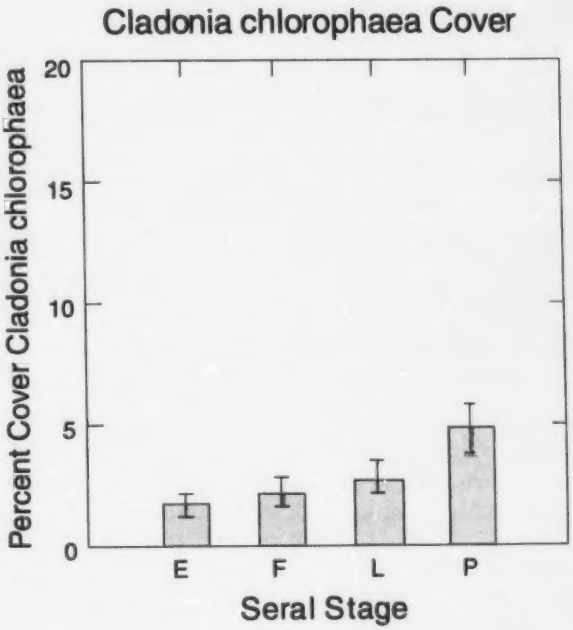

Figure 6. Percent cover of *Cladonia chlorophaea* over 4 successional stages, as described in Figure 1.

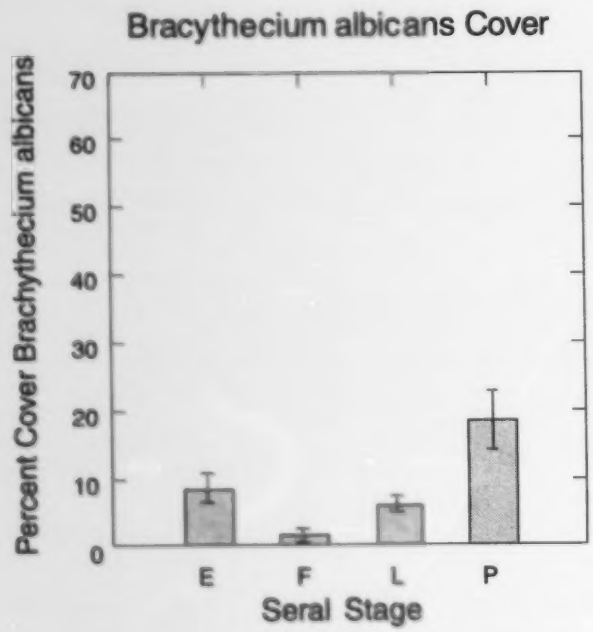

Figure 7. Percent cover of *Brachythecium albicans* over 4 successional stages.

Brachythecium albicans is most abundant in partially shaded microsites. In most stages, this pleurocarpous bryophyte is common in needle litter beneath ponderosa pine. In early seral stages it is also under *Artemesia tridentata*, while in the natural condition, where its cover is greatest, it occurs under the cover of bunchgrasses (Fig. 7).

CONCLUSIONS

This study lists 51 terricolous lichens and 39 terricolous bryophytes known to occur in British Columbia's ponderosa pine forests. This list represents the first for the lichens of this ecosystem and adds 14 species to previous reports of the bryophytes. Trends in the cover of vascular plants, bryophytes, and lichens suggest the following patterns with succession: 1) nonnative vascular plants decrease; 2) *Stipa comata* and *Cladonia pyxidata* have the greatest cover in the mid-seral stage and then decrease; 3) the large, native, perennial bunchgrasses *Festuca campestris* and *Elymus spicata*, and the lichen *Cladonia chlorophaea* increase; and 4) the pleurocarpous bryophyte *Brachythecium albicans* decreases in the mid-seral stage and has its greatest cover in the natural condition.

Analysis suggests that the surface vegetation in ponderosa pine ecosystems is composed of a dynamic community of vascular and nonvascular organisms with discernible early

and late successional species. By examining the patterns of vascular plants and microbiotic crust species it is possible to determine which seral stage a site is in. Evaluating seral development is essential to managing rangelands for an array of successional stages, and for protecting representative areas that are in a natural condition. Research in progress will elaborate on relationships between microbiotic crust species and successional patterns in this ecosystem.

ACKNOWLEDGEMENTS

Thanks to: G. Bradfield, research design, statistics, manuscript review, and supervising; M. Fairbarns, research design and logistics; S. Ford, logistics; G. Hayes, logistics; T. Goward, taxonomic consultation and manuscript review; K. Johnston, fieldwork; T. McIntosh, taxonomic consultation; T. Redding, logistics; K. Sadler, manuscript review; R. Tucker, research design and site selection; C. Wulff, manuscript review; and W. Zhang, statistics and manuscript review. Financial support was provided by the British Columbia Ministry of Forests, Range Branch.

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Disappearing Large Woody Substrate: A Habitat Change of Biblical Proportions?

Bill Chapman

Research Section, Cariboo Forest Region
British Columbia Ministry of Forests
200-640 Borland Street, Williams Lake, BC, V2G 4T1, Canada
Bill.Chapman@gems8.gov.bc.ca

Jeff Battigelli
Department of Biological Sciences
CW-405 Biological Sciences Building,
University of Alberta
Edmonton, AB, T6G 2E9, Canada

Les Paul
Department of Soil Science
University of British Columbia
139-2357 Main Mall
Vancouver, BC, V6T 1Z4, Canada

Guoping Xiao
Department of Soil Science
University of British Columbia
139-2357 Main Mall
Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

The soils of temperate and boreal forests are repositories of tremendous biological diversity. Woody debris, including boles, is one of the major energy inputs into the soil. Timber harvesting that removes boles will reduce the diversity of life in soils and make ecosystems less stable. The relationship between *Armillaria* root disease and *Hypholoma fasciculare* is presented as an example of a possible source of early stage ecosystem destabilization.

Key words: *Armillaria* root disease, biodiversity, *Hypholoma fasciculare*, woody debris.

Large woody substrates have been recognized as an important habitat component for a variety of vertebrates (Reynolds et al. 1992); however, this role is probably a severe underrepresentation of the significance of this particular habitat. Rotting wood has an intrinsic value as the sole habitat to many unique organisms. These range from the spectacular *Hericium* mushrooms, known in China as "one of the four delicacies from land and sea" (Xiao and Chapman 1996), to the lowly *Microdon* sp., a hover fly about which little is known other than that the larval stages develop in the galleries of carpenter ants.

Beyond its role as habitat, woody debris has also been identified as a site of moisture retention, free living nitrogen fixation, intense mycorrhizal activity and other ecologically significant activities (as summarized in Graham et al. 1994). We suggest that the significance of woody debris to those interested in forest management may extend beyond even those important functions. Soil is the repository of most of the diversity in temperate and boreal forests; for example, up to 500,000 individuals and 150 species/m^2 were found at the Skulow Lake Long Term Productivity Site in the Cariboo Forest Region.[1] One of the major energy inputs to natural forest soil is the boles of trees that are returned mainly intact to the soil, regardless of the type of natural disturbance.

The greatly reduced additions of fresh, large, woody material to the soil, typical of managed forests, will likely have consequences to the diversity of life in the soil; this change, in turn, will affect ecosystem stability. Specific examples of ecosystem destabilization resulting from bole removal are difficult to find. However, we think that fluctuations in the populations of certain cord-forming fungi, such as *Hypholoma fasciculare* (Huds. ex Fr.) Kummer, may directly influence the incidence of root disease such as that caused by *Armillaria ostoyae* (Romagn.) Herink and that widespread problems with *Armillaria* root disease may be a concrete example of ecosystem destabilization resulting from ecosystem simplification. *Hypholoma fasciculare* can displace the fungus causing *Armillaria* root disease from woody substrates and reports indicate an inverse relationship between the abundance of *H. fasciculare* and *Armillaria* (Wästerlund and Ingelög 1981).

We speculate that a bloom in saprophytic activity following a stand-destroying event may be important in cleansing sites of root disease organisms. Declines in populations of saprophytes and other organisms resulting from the removal of boles will likely reduce the stability of managed forests. Those forests that are managed without regard for maintaining the diversity of life within the soil will have

[1] Battigelli, J. Thesis in progress.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

increasing levels of disease and insect attack, and declining fertility and thrift; they will require increasing human inputs to maintain productivity.

SOIL BIODIVERSITY

Results collected from temperate and boreal forests around the world paint the same astonishing picture—hundreds of species, many unidentifiable, and hundreds of thousands of individuals per square metre of soil, including the forest floor. See Table 1 for an example of such data. Many of the organisms in Table 1 are known to graze fungi. In fact, sometimes beetle larvae so actively eat the cambium in roots, that they appear to eat up most of the *Armillaria*-infected cambium in stumps (pers. observation). Collembola and mites are 2 other well-known groups containing prodigious fungus grazers. While soil organisms may be small, they are often exceedingly abundant. For example, the number of earthworms found in the study reported in Table 1 worked out to be about 3,000 kg/ha. The number of bacteria and nematodes in the soil dwarf the number of arthropods and fungi but they will not be discussed further.

WOODY DEBRIS INPUTS

Many attempts have been made to quantify the levels of woody debris in forests (e.g., Means et al. 1992). In the more extreme stages of decay, levels of woody debris are difficult to quantify as log shapes may no longer be visible. Decay products, incorporated into the forest floor and the mineral soil as chemically active constituents, will still be present beyond the stage of superficially discernible woody debris. An easier number to obtain than woody debris levels, and one that may provide some meaningful insight into what is lost when boles are harvested, is how much fresh or sound woody material is removed during timber harvesting. For example, in the SBPSxc (very dry cold subzone of the Sub-Boreal Pine–Spruce biogeoclimatic zone, characterized mainly by extensive old lodgepole pine forests with little understory vegetation) where the Cariboo Forest Region Research Section has been studying the effects of timber harvesting on caribou populations, typical wood harvests are estimated to be 100 m^3/ha (H. Armleder, B.C. Ministry of Forests, pers. comm.). These yields are low compared with more productive sites where wood yields may be closer to 400 m^3/ha . However, even 100 m^3/ha of wood is the equivalent of a 1-cm thick layer of wood over the entire soil surface. Based on an approximate wood density of 400 kg/m^3, this amounts to about 8 times the annual net primary production of a grassland (5,000 kg/ha/yr; Whittaker and Likens 1973), or roughly 3 times the total biomass accumulation in a temperate grassland.

In addition to the commercially valuable wood removed from the site, tops and branches are typically piled and burned, or sometimes broadcast burnt, so that considerably more material than the harvested portion of the boles is lost to the site. The Skulow Lake Long Term Productivity site shows ranges in timber harvesting generated slash load from 10 to 40 t/ha in the Sub-Boreal Spruce (SES)dw biogeoclimatic subzone (Chapman unpublished). Kranabetter and

Table 1. Numbers of some groups of soil organisms sampled in the Sub-Boreal Spruce (SBS) dw1 (Battigelli, unpublished data).

	Max no./m^2	Avg no./m^2 over growing season
Acari	382,156	302,673
Astigmata	0	0
Cryptostigmata	184,684	161,630
Mesostigmata	15,949	10,841
Prostigmata	181,523	130,202
Araneae	338	161
Opiliones	7	2
Pseudoscorpionida	198	75
Chilopoda	13	8
Diplopoda	22	12
Collembola	125,709	65,245
Entomobryidae	2,546	979
Hypogastruridae	96,435	28,955
Isotomidae	9,399	7,343
Neelidae	7,441	1,885
Onychiuridae	36,714	17,965
Sminthuridae	587	269
Tomoceridae	2,650	1,627
Undetermined	8,681	6,233
Coleoptera (A)[a]	138	84
Coleoptera (L)	414	389
Diptera (A)	0	0
Diptera (L)	13	7
Lepidoptera (L)	9	3
Hemiptera	2,079	1,002
Homoptera	2	1
Homoptera (aphids)	7	2
Homoptera (Cicadid)	11	3
Hymenoptera	102	53
Formicidae (A)	102	51
Formicidae (L)	2	1
Formicidae (P)	20	9
Thysanoptera	98	24
Thysanura	2	1
Gastropoda	53	27
Lumbricidae	757	538
Megascolecidae	0	0
Enchytraeidae	20	9
TOTAL	551,017	370,391

[a] (A) = adult; (L) = larvae; (P) = pupae.

MacAdam (1998) report slash loads of approximately 50–120 t/ha across a range of biogeoclimatic zones. With whole tree harvesting, most of this material would be removed from the site. Kranabetter and MacAdam (1998) report that debris levels at their study sites were reduced to approximately 25–50 t/ha (roughly 50%) following broadcast burning.

Finally, the quality of wood removed by timber harvesting differs from that normally left behind after timber harvesting. Larger woody debris left after logging, standing or lying on the ground, is usually in various stages of decay. The more rapidly decomposing components of rotting wood tend to be lost in the earlier stages of decay (Means et al. 1992). Other than larger rotten wood, debris left behind includes small diameter branches and tops, and material that is often burned and that, when not burned, follows decomposition patterns considerably different from larger logs.

Studies have shown that reductions in the inputs of woody material from clearcutting, or even selective cutting, can change the populations of saprophytic fungi. Bader et al. (1995) found that higher intensities of logging activity changed substratum availability and quality (size and state of decay).

RELATIONSHIP OF *HYPHOLOMA FASCICULARE* TO *ARMILLARIA* ROOT DISEASE

The relationship between *Hypholoma fasciculare* and *Armillaria* root disease has been well documented. For example, Sokolov (1964) identified a whole series of fungi (including *H. fasciculare*) that are antagonistic to *Armillaria* root disease fungi. Sokolov recommended managing these fungi to reduce the severity of *Armillaria* root disease. In 1981, Wästerlund and Ingelög noted, "Comparison of the average abundances of *Hypholoma* and *Armillariella* [Armillaria = Armillariella] tends to show that on plots where one of the 2 species is abundant the other is less abundant." In 1990, Pearce and Malajczuk noted, "Some stumps were also colonized below ground by a naturally occurring cord-forming *Hypholoma* sp. resulting in complete or partial exclusion of *A. [Armillaria] luteobubalina* and the test fungi from lower portions of those stumps." Finally, much work has been done by the group including Dowson, Rayner, and Boddy. For example, Boddy (1993) concluded the following about cord fungi: "The ease with which they can be inoculated into the forest floor could possibly be used as a management tool by virtue of their antagonism to tree root pathogens and their abilities as agents of nutrient cycling and translocation."

Our own work has clearly demonstrated that *H. fasciculare* can outcompete *Armillaria ostoyae* in various situations. In the laboratory, *H. fasciculare* displaces *A. ostoyae* from root segments when *Armillaria* is introduced first. When both fungi are introduced simultaneously, *H. fasciculare* takes over the substrate quickly and prevents *Armillaria* from occupying it. Field work has also confirmed the superior competitive ability of *H. fasciculare* to occupy even freshly dead roots. The *H. fasciculare* moves speedily over the surface of roots via a system of cords. It appears to suppress the spread of *Armillaria* root disease under the bark, even before *H. fasciculare* has penetrated the bark. The only factor that appears to limit the ability of *H. fasciculare* to occupy a root previously occupied by *Armillaria* root disease is the inoculum potential of *H. fasciculare*.

MINISTRY OF FORESTS WORK WITH *HYPHOLOMA FASCICULARE*

We have now successfully inoculated thousands of stumps with *H. fasciculare*. We have worked with partners throughout the southern half of British Columbia, including Lignum Ltd., Crestbrook Forest Industries Ltd., Weldwood of Canada Ltd., West Fraser Timber Co. Ltd., and Slocan Forest Products. *Hypholoma fasciculare* is easy to inoculate, even in stumps previously occupied by *Armillaria ostoyae*. It grows rapidly—within 1 year from inoculation, frequently it occupies >50% of the root system. Our trials are too recent to tell if *Armillaria* root disease is being controlled. With support from the Invermere Forest District and the Invermere Enhanced Forest Management Pilot Project, we are now investigating the relationship between populations of soil fungi and the incidence of *Armillaria* root disease.

CONCLUSION

Soil in temperate forests contains an incredible diversity of fauna, fungi, and bacteria. A link appears between the health and vigour of populations of certain soil organisms, such as *H. fasciculare*, and certain forest diseases, such as *Armillaria* root disease. In fact many soil organisms are known to graze fungi and undoubtedly could influence the prevalence of root disease and many other aspects of forest health. Declining woody inputs due to timber harvesting may reduce the diversity of life in forest ecosystems and thus make them less stable. Forest managers must understand the true nature of diversity in temperate and boreal forests so that forests can be managed more efficiently to meet management objectives.

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Assessing Macrofungi of Special Concern for Conservation in Forested Ecosystems

G.A. Hunt
University College of the Cariboo
Department of Forestry and Natural Resource Science,
Box 3010, 900 McGill Road, Kamloops, BC, V2C 5N3, Canada
gahunt@cariboo.bc.ca

ABSTRACT

Species of fungi which produce conspicuous sporocarps are a major component of forest ecosystems. They are essential to ecosystem function in a number of ways and are the source of a multimillion dollar commercial industry. Many are mycorrhizal and form long-lived symbioses with the roots of plants. In this role, the fungus provides the host plant with critical soil nutrients and can increase drought tolerance as well as enhance protection against pathogens. Saprobic fungi are essential to decomposition and nutrient cycling, and pathogenic fungi affect plant health and forest stand structure. Mushrooms and truffles are also an important food source for wildlife. In spite of their economic importance and critical ecosystem functions, relatively little is known about fungal community structure, biodiversity, natural history, and rarity of species. In order to identify rare species, long-term monitoring must be conducted. To determine the fungal taxa present in a given ecosystem, both aboveground and belowground sporocarps must be monitored. In addition, mycorrhizal root tips must be assessed to identify taxa not represented in sporocarp collections. Numerous factors affect species diversity, distribution and abundance. In western North America, difficulties in assessment are compounded by diverse topography, climates, soils, and vegetation cover. At the ecosystem level, important factors include plant species composition, stand structure and age, soil type, fire, forest management practices, and commercial mushroom harvesting. British Columbia possesses a rich fungal biota. Establishing a proper monitoring system is a critical first step in identifying species potentially at risk and developing a strategy which will integrate forest fungi into ecosystem management and conservation plans.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Epigeous, Ectomycorrhizal Sporocarps in Old-Growth Forests in the Coastal Western Hemlock Very Moist Maritime (CWHvm) Zone of South Coastal British Columbia

S. Gamiet[1]
Department of Botany, University of Washington
Box 355325, Seattle, WA 98195-5325 USA1

J.F. Ammirati
Department of Botany, University of Washington
Box 355325, Seattle, WA 98195-5325 USA

ABSTRACT

Old-growth forests are constantly under pressure from logging in south coastal British Columbia. There is little information on epigeous ectomycorrhizal sporocarps specific to these ecosystems. We sampled these fungi for species richness in 2 site series each within 2 variants in old-growth forests in the CWHvm subzone. The soil moisture regimes ranged from fresh to very moist and the soil nutrient regimes ranged from very poor to very rich. Sampling occurred along a 250 X 10 meter transect, divided into 25 plots. Transects were visited every 2 weeks. Species area curves will be generated to determine the sample size required to capture most species. Results indicate that more species are found in the higher elevation montane variant than the lower elevation submontane variant. Moist and nutrient rich sites had the highest species richness compared to the fresh nutrient poor sites. Around 200 species have been identified representing 25 different genera. All species are Basidiomycetes, most species belonging to the Agaricales, Russulales and Aphyllophorales sensu Donk. A number of taxa are reported from old-growth forests in British Columbia for the first time. Some species are collected from specific site series in this subzone. Data from this study can be used to compare epigeous ectomycorrhizal sporocarps in forests that are managed to mimic old-growth attributes.

[1] Present address: 356 Defehr Rd., RR1, Abbotsford, BC, V4X 2J8, Canada
sgamiet@dowco.com

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

The Pine Mushroom: A Candidate For "Identified Wildlife" Designation Under the Forest Practices Code Act of British Columbia?

S.M. Berch

British Columbia Ministry of Forests, Glyn Road Research Station
PO Box 9536 STN PROV GOVT, Victoria, BC V8W 9C4, Canada
Shannon.Berch@gems7.gov.bc.ca

ABSTRACT

Tricholoma magnivelare (Peck) Redhead, commonly known as the pine mushroom or matsutake, is the most important mushroom commercially harvested from the forests of British Columbia. According to a recent draft Forest Renewal BC report by R. M. Wills and R. G. Lipsey, annual import of fresh pine mushrooms by Japan from BC can range from about 250,000 - 400,000 kg depending on whether it is a good year for mushrooms. At market in Japan, the value of the Canadian, largely BC, pine mushroom crop has been as high as CAN$33,000,000. As research progresses here and south of the border, we are slowly gaining information on the ecology and habitat requirements of the pine mushroom, but much remains to be determined about the factors that control its distribution and productivity. There are opportunities to manage forests for the production of timber and non-timber forest products, such as pine mushroom, but there need to be mechanisms for the identification of potential pine mushroom sites and their integration into forest management plans. "Identified wildlife" designation offers one such mechanism under the Forest Practices Code of BC. Most of the species currently included in the draft "Managing Identified Wildlife" Guidebook are Red-listed vertebrates. The Conservation Data Centre currently does not include fungi in its database and the most vulnerable fungal species have yet to be identified or assigned a rarity rank. Although the pine mushroom in BC would probably be considered a Yellow-list species (i.e., not "at risk"), it is a species that requires special management attention because it is commercially harvested virtually everywhere that it occurs in the province.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Diversity of Ericoid Mycorrhizal Fungi on *Gaultheria shallon*

T.R. Allen

Department of Botany, University of British Columbia
3529-6720 University Blvd., Vancouver, BC, V6T 1Z4, Canada
trallen@mail.botany.ubc.ca

S.M. Berch

British Columbia Ministry of Forests, Glyn Road Research Station,
PO Box 9536, Victoria, BC, V8W 3E7, Canada

M.L. Berbee

Department of Botany, University of British Columbia
3529-6720 University Blvd., Vancouver, BC, V6T 1Z4, Canada

ABSTRACT

The diversity of fungi in the field cannot be determined by looking at fruiting bodies alone. Fruiting body production varies from season to season and site to site. Small or inconspicuous fruiting bodies are easily overlooked. Many soil fungi do not fruit in the field and rarely do so in culture. So, how does one determine the diversity of fungal communities without depending on the unpredictable production of fruiting bodies? We explore here a molecular approach to determining fungal diversity. Ericoid mycorrhizal fungi are slow growing ascomycetes associated with the roots of ericaceous plants that provide their plant partners with nitrogen in nutrient poor soils. In British Columbia, fruiting bodies of ericoid mycorrhizal fungi have never been found in the field and isolates of these fungi usually remain sterile. We are using molecular techniques to identify diversity of ericoid mycorrhizal fungi associated with salal, *Gaultheria shallon* Pursh. We are culturing ericoid mycorrhizal fungi from adjacent 2-mm mycorrhizal root segments. To identify the fungi, we use polymerase chain reaction (PCR) to amplify the internal transcriber regions of the nuclear ribosomal DNA cistron. We then use restriction fragment length polymorphism (RFLP) analysis and DNA sequencing to match the isolates to known fungi. In this way we can approach the question of the "real" diversity of ericoid mycorrhizal fungi. Ultimately, this approach will reveal which fungi are common and important in ecosystem function, and which fungi are rare species in need of protection.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb ,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Bryophytes and Species at Risk in the Semi-Arid Interior of British Columbia

T. McIntosh

Botany Department Herbarium
#3529-6270 University Blvd., University of British Columbia, Vancouver, BC, V6T 1Z4, Canada
ginkgo@direct.ca

ABSTRACT

Arid-land bryophytes are poorly known both taxonomically and ecologically. Investigations have revealed, however, that there are at least 10 species that are considered at risk provincially, as well as nationally (Conservation Data Center 1998; Schofield 1998). All of these species are found in the semi-arid grasslands and steppe of the south-central portions of British Columbia from the southern Okanagan through to Kamloops, and disjunct into the Fraser-Chilcotin area west of Williams Lake, with at least 3 of the species having their only known North American distributions here. In most cases, the at-risk species are either difficult to find because of their small stature or sporadic distribution, or are taxonomically problematic. Also, bryophytes, although useful as indicators of ecological conditions and important in the habitat stability of some sites, are not usually ranked high on lists of human concern. In order to maximize the effectiveness of conservation efforts, there is a growing need for a better understanding of the ecological function and taxonomy of often overlooked arid-land groups, including not only bryophytes but also lichens, fungi, and insects. It is most important, therefore, when considering the difficulties in the taxonomy of these groups, that we focus on habitat conservation as a primary and broad goal, and not, at least in some cases, species conservation, per se. This will necessitate a coordination of a variety of field scientists, including not only bryologists and entomologists, but also scientists that are making progress with the understanding and conservation of so-called "mainstream" groups such as birds and vascular plants.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Conservation Plan For the South Okanagan

Tom Slater
Environment Canada, Canadian Wildlife Service, c/o Ducks Unlimited
945A Laval Crescent, Kamloops, BC, V2C 5P5, Canada
t_slater@ducks.ca

ABSTRACT

My presentation focuses on the South Okanagan/Similkameen valleys, which contain 1 of Canada's more endangered ecosystems. I will outline a draft Conservation Plan being proposed for this area. This plan builds on the long history of conservation in the Okanagan/Similkameen valleys. Numerous programs and activities have, and continue to be, undertaken. The result has been the securing of significant pieces of habitat, successful stewardship projects, many useful studies, and an array of excellent publications.

Key words: conservation planning, Okanagan.

The last glacial period created wider, more fertile Okanagan and Similkameen river valleys, lined by benchland terraces, the home to some special habitats. These river valleys played a very significant role in the reestablishment of flora and fauna throughout much of British Columbia's southern Interior. As the glaciers retreated they became the corridor through which plant and animal communities migrated, expanding their range northward from the ice-free refuge of the southwestern United States intermountain region. Most of the plants and wildlife of the British Columbia Interior grasslands arrived through this corridor. Also, these southern valleys, in the rain shadow of the Cascade Mountains, developed a hot, dry environment ideal for desert-adapted plants and animals. Thus was formed "Canada's Desert Country," a special, fragile ecosystem not found anywhere else in Canada—truly a "National Treasure."

With the arrival of the European explorers, settlements, cattle ranches, orchards, and vineyards very quickly began to displace or damage much of the natural habitat. As a result, a considerable amount has already been lost or damaged and the remainder is under intense pressure.

A number of species dependent on this habitat are, or have the potential to become, at risk. The migration corridor is in danger of disappearing, potentially disrupting natural evolutionary process forever. What we have is an ecosystem that is:

- Special: no other place like it in Canada.
- Tiny: represents <1% of British Columbia's land base.
- Vulnerable: desert-like, extremely fragile, sensitive to disturbance.
- Threatened: continued growth and development increase pressures on remaining habitat.

Recently the Canadian Wildlife Service took the initiative to identify an approach to preserving this area (Fig. 1) that could have a long-term impact and make a real difference. Following consultations with provincial representatives and other interested groups and individuals active in the South Okanagan, the conclusion was reached that to do this successfully the entire community must work together on a single conservation plan. Preserving this ecosystem must become a part of everyday valley life. In summary, it requires a broad program, building on present activities and past successes, embraced by all, linking conservation activities through 1 cohesive plan.

For such a plan to be supported by the community, it needs to recognize that the local economy must prosper and a good quality of life must be maintained. Growth and development would be supported where it recognized and fostered the plan's goals. The following is an overview of the proposed plan. It is still under construction.

CONSERVATION PLAN OVERVIEW

Long-term vision statement: A community composed of secure natural ecosystems, healthy populations of dependent people and wildlife, a prosperous economy, and a continued good quality of life.

Three general goals are suggested:

1. Secure a diversity of healthy habitats to sustain those species already at risk, and prevent others from becoming jeopardized.
2. Increase public support and involvement.
3. Encourage and support sustainable land-use practices that conserve healthy habitat.

Specific targets will be set for the "minimum amount of healthy habitat by type." The plan will be guided by a set of principles:

- Leadership must originate with the community.

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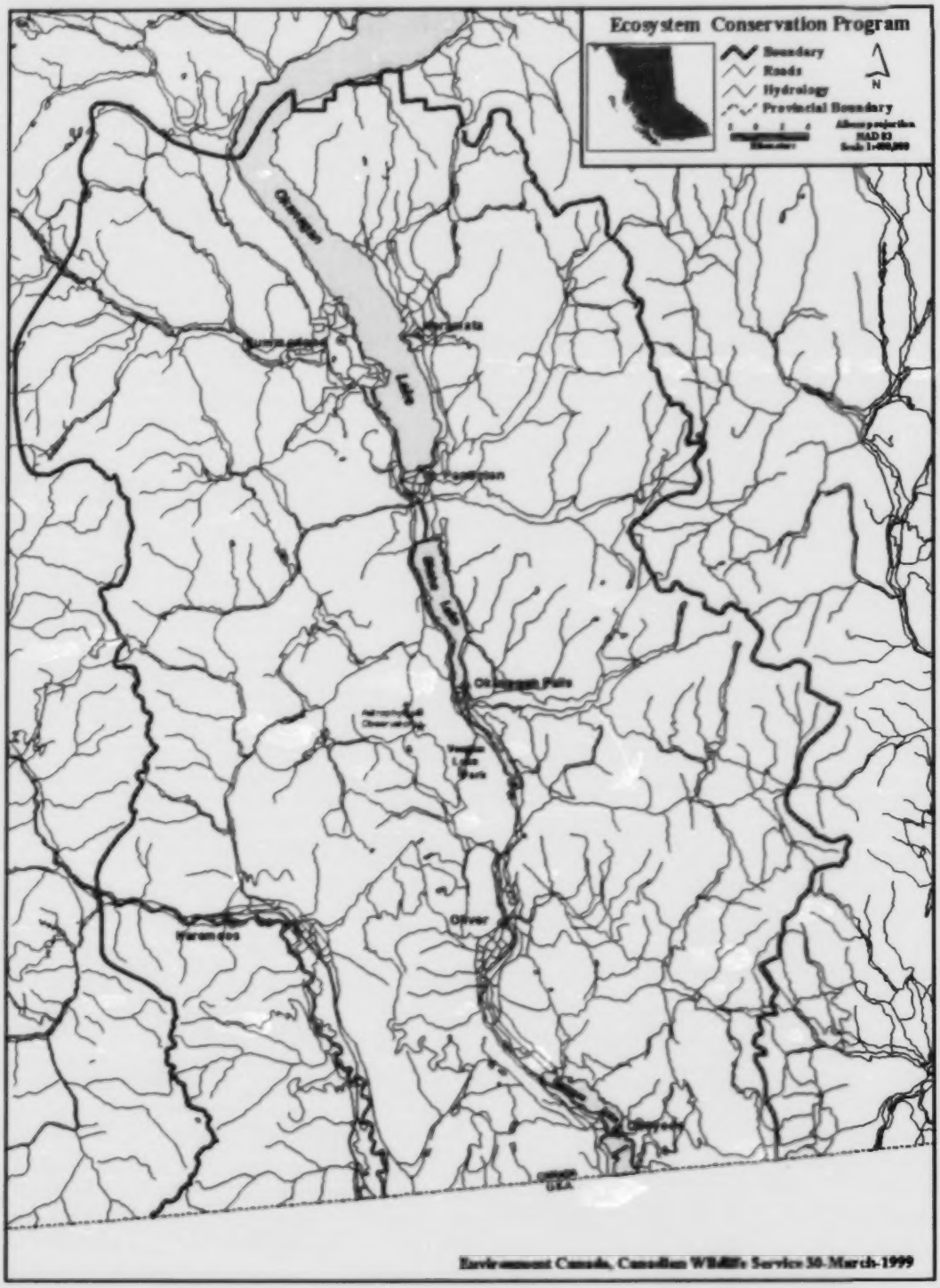

Figure 1. Boundary of the South Okanagan Ecosystem Conservation Plan.

- Large tracts of properly managed habitat must have long-term protection.
- Coordinated and cooperative efforts must exist among all levels of government and non-government interests.
- Good communication must exist between all participants.
- Plans and activities must be dynamic, adaptable, have measurable results, and be able to respond to change.
- Scientific integrity will help guide all aspects of the program.
- First Nations' rights and First Nations' participation will be recognized in all aspects of the program.
- Growth and development will be supported where it is compatible with the maintenance of healthy habitat and the species that are dependent on it.

Three independent but complimentary delivery strategies are proposed.

STRATEGY #1A: HABITAT-CONSERVATION LAND

Provide a core of secure healthy habitats on properly managed conservation land.

Actions:

- Consolidate management responsibilities.
- Create or update management plans.
- Identify additional habitat needs.

STRATEGY #1B: HABITAT-STEWARDSHIP

Partner with landowners and land managers to maximize the healthy natural habitats on their land, while maintaining their economic opportunities from this land.

Actions:

- Provide extension services to landowners.
- Support the formation of community stewardship programs.
- Expand the program to control noxious weeds.
- Develop incentives and recognitions to encourage landowner participation.
- Seek the best legal protection for participating habitats

STRATEGY #2: COMMUNITY OUTREACH

Create an informed public, supportive of the program and its goals, who are actively involved in delivery.

Actions:

- Provide local residents the opportunity to be informed and to participate.
- Increase visitor appreciation of their special surroundings
- Involve students through their curriculum.

STRATEGY #3: CONSERVATION LAND-USE

Encourage decision-makers, planners, and developers to follow practices that protect the natural healthy habitats and the species they support.

Actions:

- Identify and provide tools, techniques, and knowledge that maximize conservation.
- Work with public sector agencies to ensure their policies and guidelines promote secure, natural healthy habitats.

The local community will provide leadership through a non-government Conservation Council. Although this is not a government-led program, government support is critical. Government agencies, at all levels, must be actively involved with the community in the development and delivery of the strategies. Each strategy will be guided by a steering committee containing both government and non-government representatives. These committees will be actively involved in developing detailed action plans for achieving their objectives. When consolidated, these action plans will form the program's strategic plan.

The South Okanagan Ecosystem Recovery Team was formed as part of the National Strategy for the Recovery of National Endangered Wildlife (RENEW). The team will provide scientific guidance to all aspects of the program, monitor all research and evaluation activities, and ensure the program has the best available information and expertise. Also, it will ensure that the needs of species at risk are addressed by providing the minimum habitat requirement targets.

First Nations, individuals, and governments must have active participation at all program levels through cooperative opportunities and partnerships.

The action plan for each of the 4 strategies will determine the resources required and identify potential sources and partnerships. The plan will be delivered through existing and new initiatives, programs, and partnerships. A well-organized corps of active volunteers is integral to its success. As I mention earlier this is a project under construction.

WHERE ARE WE AT IN THIS PROCESS?

- The plan will be presented in the form of a prospectus. A draft will be circulated for comments in March[1].
- The preparation of the action plans for the Stewardship and Conservation Lands Strategies are about to begin.
- Land-use Strategy: A major tool, using computer modelling, which will help predict the long-term consequences of land-use activities on the landscape, is being developed.
- Discussions are underway on forming the Conservation Council.

The future of this special ecosystem depends on our will and our firm commitment to protect it. This is a long journey, but necessary, lest we lose our quality of life and an irreplaceable part of our heritage.

[1] This prospectus for the conservation of the South Okanagan, entitled *Conserving Canada's Desert Country*, will be published in the fall of 1999.

The South Okanagan Valley: A National Treasure at Risk

Richard J. Cannings
South Okanagan Recovery Team
1330 Debeck Road, S11, C96, R.R.#1, Naramata, BC, V0H 1N0, Canada
cannings@vip.net

Key words: *Artemisia tridentata*, antelope-brush, bunchgrass, Okanagan, ponderosa pine, *Purshia tridentata*, sagebrush.

EXPANDED ABSTRACT

The South Okanagan Valley is one of the 4 most endangered ecosystems in Canada. Its chain of large lakes moderates winter temperatures, allowing species typical of the Columbia Basin and Great Basin to extend north into Canada. The proximity of valley-bottom wetlands to arid benchlands, and the presence of montane forests that quickly merge with subalpine and eventually alpine zones, produce a mosaic of very different habitats. As a result, the Okanagan has a unique flora and fauna from a national perspective, as well as a tremendously diverse biota.

The unique characteristics of this ecosystem are centred on the low-elevation grasslands and dry forests, which are also the habitats most threatened by agricultural and urban developments. The Canadian ranges of 8 vertebrate species are restricted to the Okanagan Valley, and a further 6 species are restricted to the valley in their British Columbian ranges. Of the total Canadian breeding bird fauna, almost half (187 species) is known to nest in the South Okanagan, and the valley's 15 species of bats represent the highest bat species diversity in the country as well. The level of breeding bird diversity in the South Okanagan is matched in very few places in North America. Invertebrate diversity is far higher, of course, and 69 species have their Canadian or British Columbian ranges restricted to the South Okanagan. Seven of these species were described from the South Okanagan and have been found nowhere else in the world.

The dry grasslands and open pine forests of the Okanagan serve as a vital landscape corridor between the shrub-steppe habitats of the Columbia Basin in Washington and Oregon and the grasslands of the Thompson and Nicola valleys to the north and west. The low-elevation Bunchgrass biogeoclimatic zone makes up <33% of the study area, but 81% of the agricultural and urban development has taken place there. Less than 40% of the grasslands and shrub-steppe habitats of the Bunchgrass zone remain intact, and only 9% are in a truly natural state.

The shrub-grassland ecosystems of the South Okanagan can be divided into 3 types that differ floristically and in terms of the fauna they support: antelope-brush (*Purshia tridentata*), sagebrush, and bunchgrass. The antelope-brush ecosystem is mostly restricted to low elevations on the east side of the valley south of Penticton, usually on sandy or gravelly soils. It is the most endangered of the 3 ecosystems. Most of the present threats come from the rapidly expanding wine industry, which uses antelope-brush as an indicator species for high-quality vineyard land. Antelope-brush habitats are used by a number of Red-listed species, including the lark sparrow (*Chondestes grammacus*) and the pigmy short-horned lizard (*Phrynosoma douglasi*), which is probably extirpated from the area.

Sagebrush ecosystems are more widespread, both in the Okanagan and southern British Columbia as a whole. Nevertheless, the Canadian range of the sage thrasher (*Oreoscoptes montanus*), listed as Endangered by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC), is restricted to big sagebrush (*Artemisia tridentata*) ecosystems in the South Okanagan, as is the Blue-listed Brewer's sparrow (*Spizella breweri*). Merriam's shrew (*Sorex merriami*) is known from Canada from only 1 sagebrush site near Kilpoola Lake west of Osoyoos (Dave Huggard, University of British Columbia, 1999, pers. comm). The Canadian range of Lyall's mariposa lily (*Calochortus lyallii*) is restricted to a few spots near Kilpoola Lake as well. Pure bunchgrass habitats are not so common in the South Okanagan. The Red-listed grasshopper sparrow (*Ammodramus savannarum*) is found in these habitats in both the South and North Okanagan.

The ponderosa pine (*Pinus ponderosa*) forests of the South Okanagan also harbour a number of species of interest. These open woodlands are the habitat of white-headed woodpeckers (*Picoides albolarvatus*), which are restricted to the South Okanagan in Canada and listed as Threatened by COSEWIC. Large pines in open grassland habitats are favoured as nest and perch sites for the Blue-listed Lewis's woodpecker (*Melanerpes lewis*).

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Lewis's woodpeckers are also found in riparian ecosystems, another of the important habitat types in the South Okanagan. Riparian thickets in the valley bottom provide habitat for yellow-breasted chats (*Icteria virens*); the British Columbia chat population likely consists of <20 pairs and it is listed as Threatened by COSEWIC. Wet meadows at the north end of Osoyoos Lake are one of the most important breeding sites in British Columbia of the Blue-listed bobolink (*Dolichoryx oryzivorus*). More than half of the Canadian population of the Great Basin spadefoot toad (*Spea intermontana*), listed as Vulnerable by COSEWIC, is found in the South Okanagan, particularly in the Osoyoos Lake area.

The rugged rock cliffs and talus slopes that typify the South Okanagan also have an impressive list of species of concern associated with them. The night snake (*Hypsiglena torquata*) is found in Canada only in the Okanagan south of Penticton, as is the canyon wren (*Catherpes mexicanus*).

With the human population of the South Okanagan projected to increase to 112,000 by 2021, from its present level of about 70,000, and with increasing pressures from agricultural developments, the area has been a focus of conservation efforts for the past decade and more. Very few areas have any form of protection. The Canadian Wildlife Service manages the Vaseux-Bighorn National Wildlife Area around Vaseux Lake; Okanagan Mountain Provincial Park protects some rugged habitat at the northern end of the area; and the Nature Trust of British Columbia has purchased a number of properties, particularly in the Vaseux Lake area. The Okanagan-Shuswap Land and Resource Management Plan process, which is now nearing completion, may recommend the establishment of further protected areas on Crown land in the South Okanagan.

Most critical habitats, about 61% of the total, are in private hands or on Indian Reserves, so perhaps the most important process now underway is the South Okanagan Similkameen Stewardship program. This program provides information on the special nature of local habitats and management recommendations for landowners. Programs such as this need to be expanded and given ongoing funding if an effective ecosystem recovery process is to succeed. The South Okanagan Ecosystem Recovery team, formed under the auspices of the federal RENEW program for the recovery of endangered species, is providing scientific and technical support for this planning process, but a truly local initiative is needed as well. The Okanagan Similkameen Conservation Alliance, formed to take over from the South Okanagan Conservation Strategy, is attempting to fill this need.

Habitat Atlas for Wildlife at Risk in the South Okanagan

T. Ethier

B.C. Ministry of Environment, Lands and Parks
201-3547 Skaha Lake Road, Penticton, BC, V2A 7K2, Canada
Tom.Ethier@gems9.gov.bc.ca

M. Holm

1330 Debeck Rd., S11, C96, RR#1, Naramata, BC, V0H 1N0, Canada

ABSTRACT

In 1990, with increasing concern over the rapid loss and fragmentation of habitats in the South Okanagan, provincial and federal environment ministries, along with non-government agencies and foundations, established the South Okanagan Conservation Strategy (SOCS), a five-year program designed "to prioritize management activities for the conservation of natural habitat." The Wildlife Habitat Atlas for the South Okanagan is an initiative of SOCS, coordinated by the British Columbia Ministry of Environment Lands and Parks, Penticton office. The atlas, as well as a web-site, is designed to provide information on wildlife at risk and their habitats in the South Okanagan. In the past, information about wildlife was not easily available. This lack of information has often meant that land use decisions were implemented without consideration for the habitat requirements of wildlife species. The atlas is a distillation of many years' work in habitat inventory and ecosystem mapping in the south Okanagan and lower Similkameen valleys. Using Terrestrial Ecosystem Mapping (TEM), a large database has been generated detailing ecosystems and potential wildlife use within them. The atlas is a tool for local governments and planners to use in making land-use decisions and in designating environmentally sensitive areas requiring Impact Assessments for proposed developments. The atlas focuses on 35 species considered "at risk" in the South Okanagan-Similkameen and designated by the provincial government as endangered, threatened or vulnerable to drastic population decline or disappearance in our province. Part I of the atlas presents an assessment of the South Okanagan and how past human activity has affected our valley's ability to support wildlife within a healthy ecosystem. The maps and information show where "at risk" wildlife species can live, breed, and hibernate, along with the geographic distribution of these wildlife habitats. The information in 35 species profiles and accompanying maps can help determine the extent of habitat needed for conservation. Part II concentrates on the opportunities we have for conservation and stewardship on public and private land, from provincial conservation initiatives, to land use regulations, community stewardship, and efforts by individual landowners. We hope this atlas will be used by everyone involved in land use decisions from homeowners to planners, developers and people in local government.

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The Osoyoos Desert Society: Experimental Studies on Ecological Restoration of the Shrub-Steppe Habitat

G.G.E. Scudder
Department of Zoology and Centre for Biodiversity Research, University of British Columbia
6270 University Boulevard, Vancouver, BC, V6T 1Z4, Canada
scudder@zoology.ubc.ca

ABSTRACT

The antelope-brush ecosystem of the South Okanagan, with its many rare or potentially rare, endangered, and vulnerable species, is one of the most endangered ecosystems in Canada. Over half of this shrub-steppe habitat has been totally destroyed, with more disappearing every year. Only 9% of the remaining antelope-brush ecosystem is in a relatively undisturbed state. Alien species, especially weed invasion, livestock grazing, and other disturbance, now impact the remnants of this desert community. Little has been set aside in protected areas, and those areas now protected are too small, too few, and too scattered for effective biodiversity conservation. If current initiatives are successful, an ecosystem renewal plan for the South Okanagan will soon be established with biodiversity conservation core areas, corridors, and buffer zones, in a sound and community-acceptable landscape network. This conservation plan, focused especially in the antelope-brush community will require future emphasis on ecological restoration, habitat renewal, and management. The methodologies for these in the antelope-brush habitat have not been established. The Osoyoos Desert Society, whose objective is the saving of this desert community, has initiated a scientific and experimental study of the techniques for ecological restoration in this shrub-steppe habitat. The plans, design, and progress in the establishment of the 300 experimental plots are described, and future objectives for a desert ecosystem interpretive program and centre on the study site near Osoyoos are discussed. This interpretive program will emphasize the significance, values, and threats to the South Okanagan, and will be a vital part of the community outreach and public awareness strategy for the ecosystem.

Key words: ecological restoration, experimental studies, Okanagan, Osoyoos Desert Society, shrub-steppe habitat.

The Osoyoos Desert Society is dedicated to saving the remnants of Canada's pocket desert—the hot, dry, and fragile ecosystem confined to the southern Okanagan Valley. The society has recently acquired land near Osoyoos, on which it has started scientific experiments on ecosystem restoration, and will soon launch a public education and nature interpretation program. In this paper I summarize the ecosystem context for these studies, and then outline the restoration research and the objectives for the planned interpretive centre.

ECOSYSTEM CONTEXT

The antelope-brush ecosystem in the South Okanagan is 1 of the 4 most endangered ecosystems in Canada (Scudder 1980, Durance 1992, Schluter et al. 1995). It is restricted to the lower dry areas of the valley and is part of the vital landscape corridor between the dry grasslands and shrub-steppe habitats of the Columbia Basin, and similar habitats in the Thompson and Nicola valleys to the north and west (Scudder 1997).

Over the past century, more than 60% of this antelope-brush ecosystem has been destroyed, and converted to vineyards, orchards, croplands, or urban residential and industrial areas (Schluter et al. 1995). Much of the remainder has been disturbed and invaded by alien plants such as diffuse knapweed (*Centaurea diffusa*) and cheatgrass (*Bromus tectorum*; Cannings et al. 1999, Scott 1999). Less than 9% of the ecosystem now remains relatively undisturbed (Redpath 1990), in isolated patches on hillsides.

Less than 5,000 ha of the antelope-brush ecosystem was left in 1995 (Schluter et al. 1995), most of this in private ownership or on Indian Reserves. About 60% was in 1 area, namely the Osoyoos Indian Reserve. Over the next 5 years, Vincor International Inc. plans to convert approximately 810 ha (2,000 acres) of this shrub-steppe habitat into grape production, thus developing 1 of the largest vineyards in Canada (Hemmera Resource Consultants 1997). The first

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phase started in 1998 and aimed to plant approximately 203 ha (500 acres), and there are plans for an equivalent area to be converted in 1999 (Hemmera Resource Consultants 1997). Additional areas involving more than 100 ha of antelope-brush in East Osoyoos and other areas of the east bench were also converted to vineyards by other wine producers in 1997. The ecosystem is thus disappearing at an ever-increasing rate.

Very little of the antelope-brush steppe is preserved within officially designated protected areas (Scudder 1993, Schluter et al. 1995). Two small ecological reserves north of Osoyoos (the Field's Lease at 4 ha, and the Haynes Lease at 101 ha) protect token tracts (Scudder 1993). The latter was almost entirely burned in 1993 (Schluter et al. 1995). Although there are some additional areas of antelope-brush protected to some degree in provincial Wildlife Habitat Reserves, the Vaseux Bighorn National Wildlife Area, and areas owned by The Nature Trust of British Columbia around Vaseux Lake, these in total are too small, too few, and too scattered for effective biodiversity conservation (Scudder 1993).

This shrub-steppe ecosystem is vital for survival of many rare species in British Columbia (Scudder 1980, 1992, 1993, 1997; Durance 1992; Cannings and Cannings 1995; Schluter et al. 1995; Cannings et al. 1999). The ecosystem provides critical habitat for 30% of the Red-listed and 46% of the Blue-listed vertebrates in the province, with several of these listed as nationally Threatened or Endangered by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC; Cannings et al. 1999). There are many rare vascular plants whose range in British Columbia is restricted to this area in the South Okanagan (Cannings et al. 1999). More than 24 invertebrates are known from only this small area of the world, with an additional 80 species occurring here and nowhere else in Canada (Scudder 1992, 1994, 1996; Cannings and Cannings 1995).

To save these species, their habitats must be conserved. However, with so little of the ecosystem now represented in protected areas, with so much pressure now to develop this "raw" or "useless" land, and with the anticipated future doubling of the human population in the valley by the year 2021 (Cannings et al. 1999), it is obvious that some major conservation initiatives are needed immediately.

The antelope-brush ecosystem lies within the area being considered by the Okanagan/Shuswap Land and Resource Management Plan. This planning process is currently in a draft phase, and it is to be hoped that the remnants of this endangered shrub-steppe will receive priority for conservation in some way. An ecosystem recovery plan is being developed (Cannings et al. 1999), and this will be a major component of the upcoming "Integrated Landscape Planning and Assessment" program for the South Okanagan (Olson + Olson Planning and Design Consultants 1998).

The essential plan for an integrated landscape conservation network, with core areas, buffer zones, and connectivity corridors, has been outlined (B. C. Ministry of Environment, Lands and Parks 1999). It is obvious that ecological restoration and habitat renewal will be a priority for such landscape planning. Until now, the required ecological research on habitat restoration in the antelope-brush community has not been undertaken. This research is now underway.

EXPERIMENTAL STUDIES ON ECOLOGICAL RESTORATION

The Osoyoos Desert Society, through its ad-hoc Scientific Committee, has received funding from the Habitat Conservation Trust Fund for research on "Ecological Restoration and Habitat Renewal of the South Okanagan Shrub-steppe." With additional financial support from Environment Canada, and Human Resources Development Canada, detailed comparative experiments are now underway at the Osoyoos Desert Society site near Osoyoos.

This 26.8-ha site, acquired by the Regional District of the Okanagan-Similkameen, through the cooperation of Crown Lands, the British Columbia Ministry of Forests, the Town of Osoyoos, the Osoyoos Indian Band, and rancher J. Pendergraft, is now leased to the Osoyoos Desert Society. It contains 2 provincially Red-listed shrub-steppe communities, namely the antelope-brush (*Purshia tridentata*) and the Great Basin sage (*Artemisia tridentata*) associations. When first assessed in 1996, the site was heavily grazed, and was considered to be in fair condition, but the trend was for continued deterioration because of a high percentage of non-indigenous weedy species, a low percentage of perennial native grasses and forbs, and a very low vascular plant recruitment rate (Atwood 1996). This assessment was made possible through the cooperation of J. Pendergraft, who also allowed the invertebrate diversity on the site to be determined while still subject to livestock grazing. This invertebrate study emphasized the ground-dwelling species, and was sampled by pit-fall trapping in 1996–97, with Forest Renewal British Columbia support.

The aims of the restoration research are: 1) to restore the community structure, function, and species composition of the antelope-brush ecosystem, and 2) to develop effective and economical restoration techniques for degraded natural areas of this ecosystem. The adjacent Field's Lease Ecological Reserve will serve as a benchmark reference.

In the research, experiments are being conducted in paired 100-m^2 experimental and control plots, with each treatment replicated 5 times. Some 30 experiments are planned. Thus, >300 plots were established in 1998, and characterized with respect to plant diversity, and vegetation and soil condition. Control plots and other areas not used for the experimental plots will be studied to assess the response

to removal of livestock grazing.

Planned experiments can be classified as either (a) passive methods for ecological restoration, or (b) active methods for ecological restoration. Passive methods to be investigated include: 1) effectiveness of mechanical (cutting, pulling, or burning) and chemical control of Dalmation toadflax (*Linaria genistifolia* var. *dalmatica*) and diffuse knapweed; 2) effectiveness of using various herbicide application rates and timing for control of annual brome or cheatgrass; and 3) effectiveness of solarization for weed control. The latter, which is achieved by covering experimental plots with clear plastic (kindly proved by K & P Silviculture, Oliver, B.C.), is expected to encourage germination, growth, and, subsequently, death from overheating. The mechanical control experiments (excluding burning) with Dalmation toadflax and diffuse knapweed, as well as the solarization experiments, were started in 1998.

Experiments using active methods of restoration will be complex, and will attempt to (a) determine the most effective plant materials and methods of planting, and (b) determine the most beneficial soil manipulations and modifications for plant establishment. Most of these experiments will not start until the year 2000, because they depend on the use of native seed that is being collected from surrounding areas during the first 2 years of the research. However, some hay-seeding experiments were started in 1998, and additional experiments involving manipulations of the native microbiotic crust and the soil mycorrhizal fungi are planned for 1999.

Weed control in areas surrounding the experimental plots has started, alien shrubs and trees (mostly Siberian elm [*Ulmus pumila*]) have been removed, and pruning has been done to open shrub and riparian canopies. Considerable effort is being made to reduce all other impacts on the site, a major concern in such a small area under intensive investigation for a long period of time. In 1998, >3,000 kg of weeds was removed, along with many alien metal and glass objects.

Ecological restoration is a long-term project, with few starting results to be expected in the near term. The alien weed-seed bank in the soil is likely to persist for >15 years, and this is the time frame for realistic assessment of this research.

The experimental studies, under the direction of L. Atwood of Genoa Environmental Consulting, will continue over the initial 5-year experimental plan, provided outside funding is continued. At the same time, I will continue monitoring the response of the ground-dwelling invertebrate community to the various changes and manipulations. This research is currently funded by grants from the Natural Sciences and Engineering Research Council of Canada.

PUBLIC EDUCATION AND INTERPRETIVE PROGRAM

The Osoyoos Desert Society is planning for a desert ecosystem interpretive program and centre on the study site. A 3-km walkway around the site has been established, and a wooden boardwalk will be constructed in 1999. Once this is completed, by early summer, a public interpretation program will be launched.

Initially, this will concentrate on educating local citizens and visitors about the ecosystems of the South Okanagan, especially the antelope-brush ecosystem and its essential components, characteristics, and threats. The ongoing habitat restoration research will be explained and related to the long-term plans for ecosystem renewal and a comprehensive landscape conservation network. The basic concepts of conservation biology will be explained, with reference to the rationale behind the establishment of core conservation areas, buffer zones, and corridors. Particular emphasis will be placed on the importance of stewardship and public involvement.

The Society hopes to construct the first module of the interpretive centre by the year 2000. It plans for this to be a major centre for school education programs, and a popular attraction in the area, where the public can obtain details of the values and threats to the fragile ecosystems of the South Okanagan. Particular emphasis will be put on local public involvement, because, in the long run, this is seen as the most important component for saving the many species at risk in these endangered ecosystems.

ACKNOWLEDGEMENTS

I am indebted to L. Atwood, P. Krannitz, J. Miller, and L. Scott for their help and input into the restoration program. Other vital support has been noted in the text. The research would not have been possible without the encouragement of M. Clapson, L. Dallas, F. Hamilton, and R. Schiller.

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The Legacy of Private Lands: Acknowledging the Landowners

A.D. Bryan

South Okanagan Similkameen (SOS) Stewardship Program
c/o # 201-3547 Skaha Lake Rd., Penticton, BC, V2A 7K8, Canada
abryan@vip.net

S.M. Austen

South Okanagan Similkameen (SOS) Stewardship Program
c/o # 201-3547 Skaha Lake Rd., Penticton, BC, V2A 7K8, Canada

ABSTRACT

Conservation of wildlife species and plant communities at risk in the lower elevations of the south Okanagan-Similkameen depends largely on the support of private landowners. For example, private land and Indian Reserves support 84% of the remaining Antelopebrush - Needle-and-thread community and 88% of the Black cottonwood - Water birch community. Stewardship provides the opportunity to raise awareness, identify areas of mutual benefit, build partnerships, and facilitate habitat enhancement and long-term securement. Many landowners in the south Okanagan-Similkameen have a strong respect for, and knowledge of, their land and the wildlife it sustains. Often, all that is needed is encouragement and support in the form of integrated information and technical assistance. Since 1994, the South Okanagan-Similkameen (SOS) Stewardship Program has played a pivotal role in the South Okanagan Conservation Strategy. This proactive program encourages the conservation and enhancement of Red- and Blue-listed plant communities and wildlife species on private lands, especially those adjacent to conservation lands, in a variety of ways. These include providing information on habitat values, wildlife species, and environmentally sustainable land management practices; building trust through person-to-person contact with landowners of large holdings; preparing management recommendations; assisting with selected conservation projects; recognizing stewards through award ceremonies; and facilitating long-term habitat securement. Partnerships between landowners and SOS Stewardship, as well as other conservation organizations, have resulted in a variety of conservation projects that serve to demonstrate good stewardship. However, there is an urgent need to reach a broader spectrum of landowners and the public. Specific community stewardship projects to address this need will be discussed.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Ecological Restoration and Habitat Renewal of the South Okanagan Shrub-Steppe

Lynne B. Atwood
Genoa Environmental Consulting
Box 20282, Smithers, BC, V0J 3P0, Canada
latwood@bulkley.net

Osoyoos Desert Society
Box 123, Osoyoos, BC, V0H 1V0, Canada

ABSTRACT

An ecosystem recovery project is beginning in a 50-ha shrub-steppe site in the South Okanagan, British Columbia. The project has 2 objectives: (1) to restore the community structure, function, and species composition of the natural habitat; and (2) to develop effective and economical restoration techniques for degraded natural systems.

Key words: bunchgrass, ecosystem restoration, hayseeding, shrub-steppe, South Okanagan, weed control.

The South Okanagan is a biodiversity hot spot (Mosquin et al. 1995) and the shrub-steppe communities of this area are an integral part of the ecosystem. However, fragmentation and livestock grazing have degraded the native shrub-steppe plant communities and the consequences of past land use are reduced diversity and cover of native plant species, increased bare ground, and invasion by nonindigenous species. An ecosystem recovery plan is being prepared for the area and ecosystem restoration and habitat renewal are priorities for such planning (T. Slater, Canadian Wildlife Service, pers. comm.). Until now the required research on habitat restoration has not occurred. This project addresses this lack of restoration knowledge.

The Osoyoos Desert Society acquired 50 ha of shrub-steppe and began a 5-year program for ecological restoration and habitat renewal of the site in 1998. The project area is in the Southern Okanogan Basin Ecosection and the Okanagan very dry hot Bunchgrass (BGxh1) biogeoclimatic subzone and variant. Two provincially Red-listed shrub-steppe plant communities, antelope-brush–needle-and-thread grass (*Purshia tridentata–Stipa comata)* and great basin sage–bluebunch wheatgrass (*Artemisia tridentata–Elymus spicata*), occur on the project site. The listed communities are habitat for >100 rare plants, 300 rare invertebrates, 29 Red- and Blue-listed vertebrate species, and 4 species of management concern for the South Okanagan Conservation Strategy (Scudder 1994, 1996; Bryan 1996; Conservation Data Centre 1997).

OBJECTIVES AND METHODS

The ecological restoration and habitat renewal project has 2 objectives: (1) to restore the community structure, function, and species composition of the natural habitats, and (2) to develop effective and economical restoration techniques for degraded natural systems. Four project goals will guide the restoration techniques for the site: (1) reduce the weedy species component, (2) increase the cover and recruitment rate of native grasses and forbs, (3) replace areas of early successional species with climax species, and (4) maximize species diversity within each of the identified habitat types.

Restoration techniques follow 2 approaches: rigorous scientific experiments and general habitat enhancement. Both passive and active restoration methods will be tested. Experiments focus on alternative weed control measures, efficiency of restocking strategies, and effectiveness of soil amendments. Additional habitat enhancement activities that will increase community structure and plant diversity will occur in areas of the site that are not used for experimental plots.

YEAR 1: 1998–1999

Objective 1: To restore the community structure, function, and species composition of the natural habitats.

1. Collecting and cleaning native seed from 4 perennial bunchgrasses, 4 shrubs, and 1 forb.
   - Approximately 40% of the native seed required for future experiments was collected during the 1998 field season. The resources required to obtain the native seed were staggering. Extensive stands of native bunchgrasses are

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exceedingly rare in the South Okanagan and access to them is difficult. Most of the remaining natural areas are on private land or are protected and many landowners and land managers are reluctant to allow native seed harvesting. In addition to a poor supply, few seed collectors are trained, and seed cleaning techniques are rudimentary. The time required to pick and clean the native seed varied by species, but producing 1 kg of pure seed (pick and clean) ranged from 6 person-hours for sand dropseed (*Sporobulus cryptobulus*) to 57 person-hours for bluebunch wheatgrass.

2. Enhancing habitat by controlling weeds, hayseeding[1], building habitat structures, and designating walking trails.
   - 3,075 kg of noxious weeds were hand-pulled, bagged, and removed from areas around the experimental plots. The targeted species were diffuse knapweed (*Centaurea diffusa*), dalmatian toadflax (*Linaria genistifolia* ssp. *dalmatica*), hound's tongue (*Cynoglossum officinale*), and mullein (*Verbascum thapsus*).
   - Following weed removal, bare soils were covered with native hayseed.
   - Old roads and trails were deactivated. The roads and trails were blocked with piles of brush. These piles provide cover and roosting sites for wildlife.

Highly competitive weed species are colonizing the disturbed soils on the site. Weed colonization uses limited resources and prevents the establishment of native species (Ross and Lembi 1999). We are reducing soil disturbance on the site by designating walking trails; however, by removing weeds and supplying a native seed source we are also encouraging the establishment of native species. In addition to seed, native hayseeding material supplies organic matter, which helps rebuild soil structure and bolster nutrient cycling. The habitat structures also contribute to seeding of disturbed areas because reptiles, small mammals, and birds are valuable seed dispersers in the shrub-steppe system and they will use more of a site if shelter, nesting, and roosting areas are readily available.

Objective 2: To develop effective and economical restoration techniques for degraded natural systems.

1. The delineation, staking, and tagging of 305 100-m^2 experimental plots (60 plots each at 5 experiment replicates at the same study site, plus 5 plots selected solely to assess the effect of control of dalmation toadflax).
2. Collection of vegetation data from 3 1-m^2 quadrats randomly located within each 100-m^2 plot (n = 915). Percent cover, distribution and vigour of the vascular plant species, and percent cover of moss, lichen, litter, and bare ground were collected from each quadrat. Soil texture data were collected from the 305 100-m^2 plots.
   - Soil texture differed over the site. Textures in replications (Reps) 1–4 were similar, but Rep 5 soils contained more sand and less silt (Fig. 1). On average, Rep 5 also contained a higher cover of litter, bare ground, weeds, and native herbs, but fewer native grasses than Reps 1–4 (Figs. 2, 3). There was no difference in the average amount of non-vascular plant cover between areas (Fig. 4).
3. The initiation of 1 active experiment (hayseeding perennial bunchgrasses) and 4 passive restoration experiments (alternative manual weed control techniques for diffuse knapweed and dalmatian toadflax and an assessment of plant community response without restoration).
   - Stalks of native grasses were applied to the experimental plots as seed ripened. Bluebunch wheatgrass (*Elymus spicata*) was applied first, followed by needle-and-thread grass (*Stipa comata*), sand dropseed (*Sporobolus cryptandrus*), and red three-awn (*Aristida longiseta*).

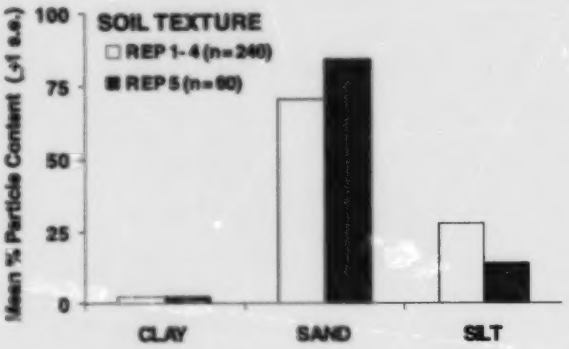

Figure 1. Average percent of clay, sand, and silt particles in Reps 1–4 soils and Rep 5 soils (mean ± 1 SE).

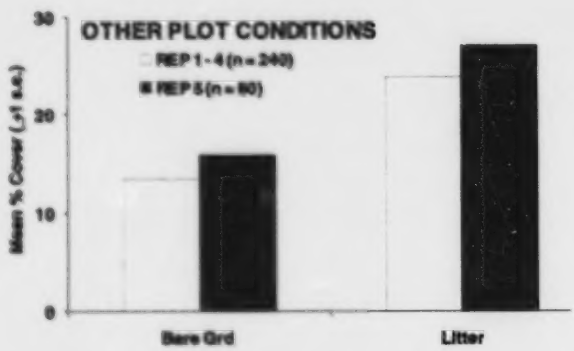

Figure 2. Average percent cover of litter and bare ground in Reps 1–4 and Rep 5 (mean ± 1 SE).

[1] Hayseeding is the mowing or cutting of plant stalks with mature seed heads from similar vegetation communities and scattering them over the planting site (Cottam 1987).

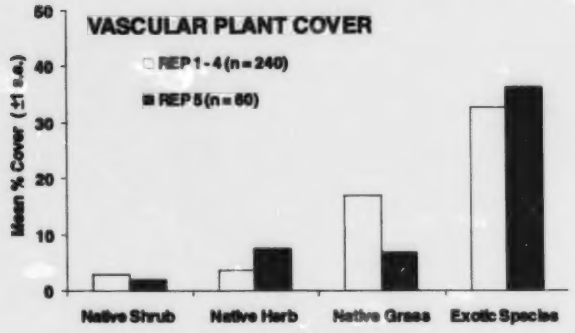

Figure 3. Average percent cover of native shrubs, herbs, grasses, and exotic (weed) species in Reps 1–4 and Rep 5 (mean ± 1 SE).

Scientific research is required to develop effective and economical techniques for ecological restoration of British Columbia's dryland habitats. In the 1998 field season, we set up the experimental plots and collected the baseline data. After data collection, 5 of the planned restoration experiments were initiated. Dr. G. G. E. Scudder (University of British Columbia) is monitoring the use of the native habitats by invertebrates. His monitoring program will determine if habitat use changes in response to the restoration treatments.

Activities Planned for 1999–2000

The 1999–2000 activities include the initiation of 2 passive and 2 active restoration experiments; monitoring and data collection for the restoration experiments initiated in 1998; continued habitat enhancement of areas surrounding the experimental plots; monitoring and possible retreatment of habitat enhancement activities completed in 1998, and carrying out preliminary activities for experiments planned for the 2000–2001 season.

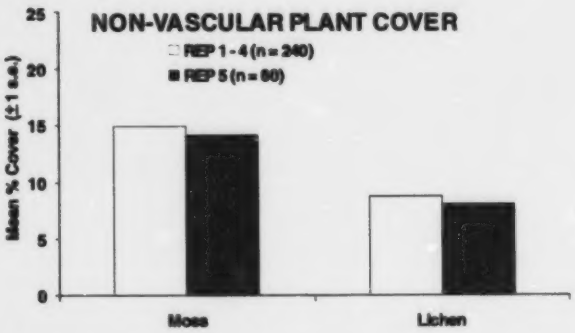

Figure 4. Average percent cover of nonvascular plants (mosses and lichens) in Reps 1–4 and Rep 5 (mean ± 1 SE).

DISCUSSION

The primary value of the restoration project is the conservation and enhancement of endangered habitat and the subsequent benefit to wildlife dependent on the natural systems. In addition, experiments are the beginning of a 5-year demonstration project on ecological restoration and habitat renewal in degraded natural systems. Results from the experiments will enable scientists and land managers to determine ecological protocols for protected areas and habitat remnants. The experimental treatments will also be a central part of the interpretative program of the Osoyoos Desert Society and will educate and involve the public in habitat preservation and renewal techniques.

The experimental results are years away, but the need for native seed has already been identified. We must develop a native seed industry that can supply British Columbia seed for British Columbia habitats. Research on seed transfer limits for native grasses and forbs, appropriate seed mixes, and efficient seed handling techniques are required—again, results would be years away. As an immediate remedy, land managers considering ecological restoration should develop native seed increaser fields. The current rate of land conversion is threatening the future supply of native seed.

ACKNOWLEDGEMENTS

I would like to acknowledge the Habitat Conservation Trust Fund, Environment Canada, University of British Columbia, Human Resources Development Commission, and the Osoyoos Desert Society for funding this restoration project. I also thank the members of the Osoyoos Desert Society Science Committee for their assistance. Committee members include chair, Dr. G. G. E. Scudder (University of British Columbia); Dr. P. Krannitz (Environment Canada); J. Millar (British Columbia Parks); and L. Scott (consultant).

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A Systematic Method for Identifying Priority Areas to Conserve Rare Species Using 'Irreplaceability': A Test Case for the South Okanagan

L. Warman

Centre for Biodiversity Research, Department of Zoology, University of British Columbia
6270 University Boul., Vancouver, BC, V6T 1Z4, Canada
warman@zoology.ubc.ca

ABSTRACT

Managers and biologists have recognised the need for systematic approaches to conserve rare species and habitats. Geographic Information Systems provide a tool that can be used to identify priority areas for conservation using geographically referenced data. A software program, called C-Plan, has recently been produced to identify conservation areas in landscapes that are subject to the effects of human development. Conservation areas are selected using complementarity, which is an approach that identifies the minimum or near minimum amount of area required to meet a conservation goal. C-Plan provides a tool for prioritizing conservation areas by calculating the irreplaceability of user-defined sites. Irreplaceability is a measure of the conservation importance of a site based on the species richness and rarity of the site and the vulnerability of the site to harmful impacts. The selection process inherent in C-Plan is efficient and repeatable, and can be used for predicting the outcome of different policy and management decisions on regional conservation goals. The South Okanagan has been chosen as a test case for the application of the software program in Canada because of the high amount of national endemism present in this area and the imminent threat of habitat loss from human habitation. This study uses Terrestrial Ecosystem Mapping as the base for the analyses, habitat distributions of 29 Red- and Blue-listed vertebrate species, and locations of rare vascular plants to evaluate options for the conservation of habitat in this region. The land tenure and the agricultural and resource values are used to identify high priority sites that are in the least conflict with competing land uses; together they aid in the realization of the regional conservation goals.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Effect of the Microbiotic Crust of the Antelope-Brush (*Purshia tridentata*) Shrub-Steppe on Soil Moisture

Lynne B. Atwood
Genoa Environmental Consulting
Box 20282, Smithers, BC, V0J 3P0, Canada
latwood@bulkley.net

Pam G. Krannitz
Canadian Wildlife Service, Pacific Wildlife Research Centre
5421 Robertson Rd., RR 1, Delta, BC, V4K 3N2, Canada

ABSTRACT

Microbiotic crusts moderated soil moisture conditions in the antelope-brush shrub-steppe of the South Okanagan and the influence was higher in areas with extensive amounts of crust. However, the life-forms in the crust played different roles.

Key words: antelope-brush, microbiotic crust, *Purshia tridentata*, shrub-steppe, soil moisture, South Okanagan.

The microbiotic crust is a complex mix of lichen, moss, liverworts, algae, fungi, and bacteria that colonize the soil surface in semiarid environments. Microbiotic crusts are also referred to as cryptobiotic (Belnap 1994), cryptogamic (Kleiner and Harper 1972), microflora' (Loope and Gifford 1972), or microphytic crusts (West 1990). Microbiotic crusts have been linked to 6 ecological functions: soil formation, soil surface stabilization, nutrient cycling, vascular plant germination and establishment, a source of food and shelter, and soil water relations (West 1990).

This study examined the role of the microbiotic crust on soil moisture and addressed 3 issues: (1) if the microbiotic crust influenced soil water content and loss; (2) if the soil moisture effect differed with microbiotic crust life-form; and (3) if the role of the microbiotic crust on soil water was more important than influences from other ecosystem components (Atwood 1998). The study area was the provincially Red-listed antelope-brush (*Purshia tridentata*) shrub-steppe in the South Okanagan.

METHODS

Eighty-eight 1-m^2 plots were established at 2 sites, Water Tower (WT) and Kennedy Bench (KB). The sites were chosen because they contained similar soil textures but different amounts of bare ground, variables that may influence soil moisture retention. The average cover of bare ground at KB was 4% compared with 22% at WT. Soil moisture was measured 1, 3, and 5 days after a rain event, in August and October, at a 15-cm^2 sample point in each plot. Half of the plots had sample points covered by microbiotic crust and half were bare. Two variables (crust and shrub) were tested in a 2 x 2 factorial design with 22 plots per combination (crust, shrub, crust x shrub, no crust or shrub [bare]).

Shrub cover was included in the experiment to account for the possible effects of hydraulic lift[1] and shading on soil moisture. Data on soil texture, percent cover of each non-vascular and vascular plant life-form, and thickness of the moss in the microbiotic crust were collected from the entire 1-m^2 plot. Soil water content data were analyzed using analysis of variance (ANOVA), moisture loss over time was analyzed with Repeated Measures multivariate ANOVA (MANOVA), and Pearson's correlation coefficient (r) was used to identify associations between soil moisture and plot conditions. The percent cover data were arc-sine square root transformed prior to analysis.

RESULTS

Soil Moisture Content

After the 8-mm August rainfall the effect of microbiotic crust on soil moisture was similar at KB and WT. Soils beneath microbiotic crust at the sample point (crusted plots) received less rainwater initially, but retained moisture longer than bare soil (non-crusted plots) (Fig. 1). Soil water content was higher in crusted plots within 3 days of the rainfall and this trend was still evident 5 days after the rain. On day 5, soils covered by microbiotic crust contained almost twice the soil moisture of bare ground.

Rain showers were light and sporadic during the October data collection. For the first 3 sampling days soils covered by

[1] Hydraulic lift is the ability of deep rooted species to bring moisture to the upper soil layers where it can be used by shallow-rooted species (Caldwell 1990).

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microbiotic crust contained more moisture than bare soils, but soil moisture conditions changed after a 2-mm rainfall on day 4. Soils covered by microbiotic crust absorbed little to none of the light shower, while soil moisture increased substantially in bare soils (Fig. 2). The 2-mm rain shower highlighted the differences in the water absorption patterns of the microbiotic crust life-forms. Soils beneath thick moss did not receive the light rain shower but soils beneath lichen crusts did (Table 1). Mosses in the microbiotic crust inhibited soil water percolation, but lichens did not.

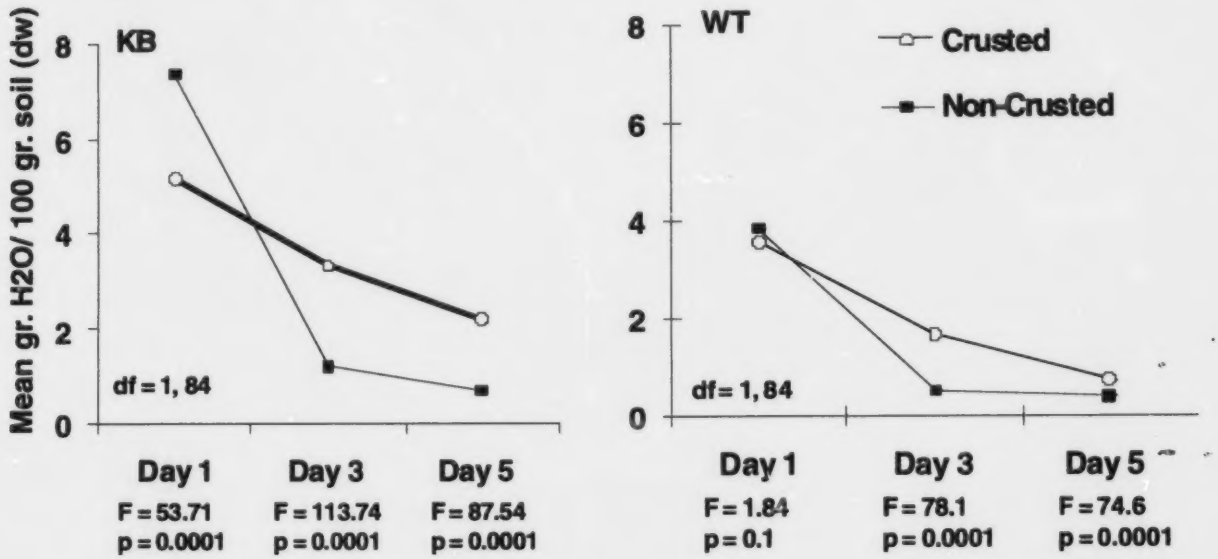

Figure. 1. Soil water content at 2 sites in crusted and non-crusted plots from day 1 to day 5 following a rain event in August (mean ±1 SE).

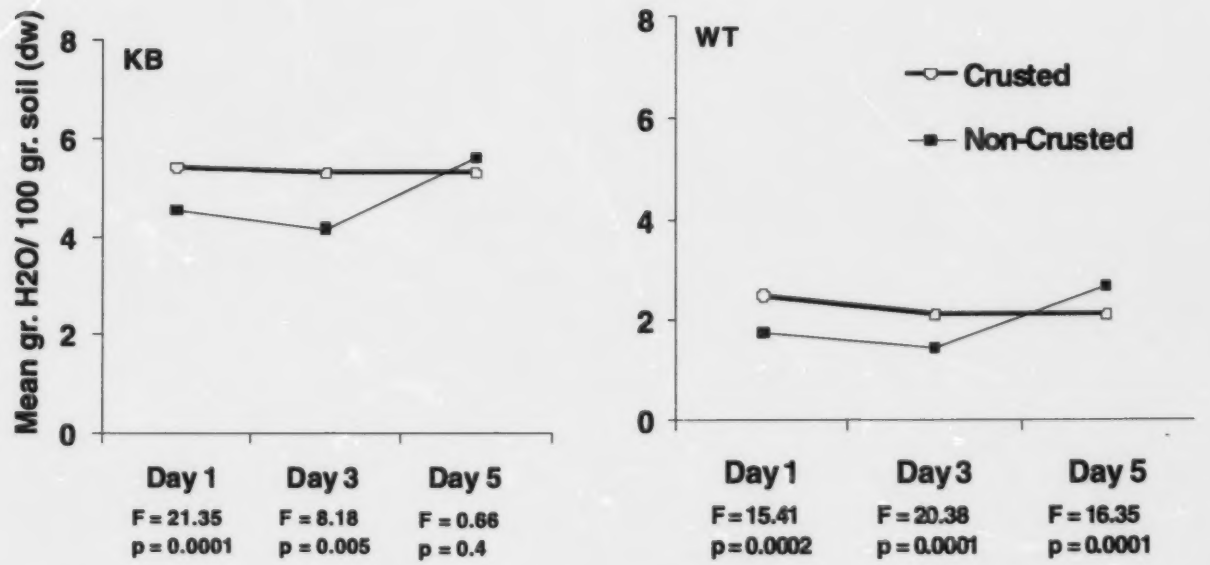

Figure 2. Soil water content at 2 sites in crusted and non-crusted plots from day 1 to day 5 following a rain event in October (mean ±1 SE).

Table 1. The correlation between soil moisture content of KB soils covered by a microbiotic crust and the percent cover of moss, lichen and thickness of the moss in the 1 m^2 surrounding the sample point.

Kennedy Bench (KB) – October, Day 5			
Association between soil moisture content in crusted soils and	*r*	*p*	*n*
% cover of moss in the microbiotic crust	-0.47	0.001	43
thickness of the moss (mm) in the microbiotic crust	-0.32	0.04	43
% cover of lichen in the microbiotic crust	0.34	0.03	43

Soil Water Loss

In August, drying patterns differed between soils covered with microbiotic crust and those that were bare (KB [time x crust]: $F = 221.3$, $p = 0.0001$, df = 1, 84; WT [time x crust]: $F = 72.56$, $p = 0.0001$, df = 1, 83). The most rapid moisture loss occurred in the first 3 days after the rain event. On average, bare soils lost 86% of the soil moisture within 3 days of the rain, but soils covered by microbiotic crust lost an average of 44% soil moisture during this same period. Five days after the rain, crusted soils retained an average of 31% of the day 1 soil water, while bare soils retained only 9.5% of the moisture measured on day 1.

Plot Conditions

Soil moisture content and loss varied with environmental conditions surrounding the sample point. The plot conditions with the most consistent influence on soil water were the amount of microbiotic crust in the 1 m^2 surrounding the sample point, the dominant crust life-form, and life-form morphology.

Three days after the August rain event soil water content increased as the percent cover of microbiotic crust in the surrounding area increased, and the amount of water that was lost was reduced if the surrounding microbiotic crust contained thick moss (Table 2). Although moss inhibited initial percolation, it contributed to the water retention ability of the crust. The positive effect of a mossy microbiotic crust was so pronounced at KB that water content in bare soils was higher if surrounded by an extensive amount of moss (Table 2). General site conditions likely contributed to this effect as the positive relationship between microbiotic crust and moisture in bare soils was not seen at WT.

Plant litter and bare soil were the only other plot variables that consistently influenced soil water conditions. Three days after the rain, soil water content was higher in plots with large amounts of litter, and water content was very low if the sample point was surrounded by large amounts of bare ground (Table 2). Plant litter was more important for water retention in areas where the microbiotic crust was thin (WT), but the benefits were short lived and much of the soil water beneath the litter disappeared by day 5 (WT: grams of water lost between day 3 and day 5 increased with litter cover; $r = 0.38$, $p = 0.0003$, $n = 87$).

WT and KB contained the same average percent cover of microbiotic crust and the influence of the crust on moisture conditions was similar. Despite receiving the same amount of

Table 2. The correlation between soil water content or the grams of water lost and conditions in the 1 m^2 surrounding the sample point for Kennedy Bench and Water Tower.

Correlation with plot variables	Kennedy Bench			Water Tower		
Association between day 3 soil water content and	r	p	n	r	p	n
% cover of microbiotic crust in the 1-m^2 plot	0.4	0.0001	88	0.36	0.0006	87
Association between the grams of soil water lost between day 1 and day 3 and						
% cover of moss in the 1-m^2 plot	-0.54	0.0001	87	-0.54	0.0001	87
moss thickness (mm) in the 1-m^2 plot	-0.45	0.0001	85	-0.51	0.0001	87
Association between soil water content on day 3 in non-crusted (bare) plots and						
% cover of microbiotic crust in the 1-m^2 plot	0.32	0.03	44	-0.08	0.6	44
% cover of moss in the 1-m^2 plot	0.5	0.0006	44	-0.12	0.4	44
Association between soil water content on day 3 and						
% cover of litter	0.27	0.01	87	0.43	0.0001	87
% cover of bare ground	-0.73	0.0001	88	-0.52	0.0001	87

rainfall, one day after the August rain, non-crusted soils at KB contained almost twice the moisture of similar areas at WT (Fig. 1). The influence of surrounding plot characteristics likely contributed to this day 1 difference. KB contained 36% more plant litter, 82% less bare ground, and mosses that averaged twice the height of WT mosses; characteristics found to benefit soil water. It is likely the sparsely covered soils at WT dried rapidly, losing a large amount of the rain water before the day 1 collection.

DISCUSSION

Microbiotic crusts moderated soil moisture conditions in the antelope-brush shrub-steppe of the South Okanagan and the influence was higher in areas with extensive amounts of crust. However, the life-forms in the crust played different roles. Lichens were related to increased soil water absorption, while mosses were associated with prolonged soil moisture retention. In areas of sparse or thin crust, plant litter moderated moisture conditions, but the effect at conserving soil moisture was short lived. The cover of plant litter at WT averaged 57%, but over one-half of the moisture received during the August rainfall was lost before the day 1 collection. The extensive amount of bare ground on the site and the thin mosses could not retain the soil water.

The prolonged retention of soil moisture in semiarid environments is critical for seedling survival, plant growth, reproduction, and plant persistence (Dasberg 1971, MacMahon and Schimpt 1981, Lesica and Shelly 1992, Gutterman 1993). An extensive, diverse, and well-developed microbiotic crust would contribute to the colonization process of antelope-brush communities; however, crust development takes time. Restoration efforts that increase the recruitment of microbiotic crust species would accelerate development of both the nonvascular and vascular plant communities of the Red-listed, antelope-brush shrub-steppe.

ACKNOWLEDGEMENTS

We thank the landowners of the South Okanagan who allowed access to their land during this study and the Habitat Conservation Trust Fund, Endangered Species Recovery Fund, Environment Canada, and UBC Graduate Fellowship program for contributing to this project.

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Weed Invasion in the South Okanagan

L. Scott
RR #3, Site 38, Comp. 24, Summerland, BC, V0H 1Z0, Canada
lscott@vip.net

D. Ralph
British Columbia Ministry of Agriculture and Food
162 Oriole Rd., Kamloops, BC, V2C 4N7, Canada
David.Ralph@gems5.gov.bc.ca

ABSTRACT

Plant invasions are a serious global threat to natural and managed habitats. In British Columbia, over 20% of the vascular plants have been introduced. Weeds are adapted to a wide range of environmental conditions but occur most often in disturbed soils and depleted rangelands. They compromise land values for wildlife, livestock, agricultural crops, recreation and aesthetics. Prolific seed production, varied dispersal mechanisms and an ability to tolerate severe stress conditions contribute to the ecological success of weeds and their often rapid colonization of habitats. The current assessment focused on several plant families and included terrestrial, wetland and aquatic plants. The earliest records for these species was based primarily on information gathered from herbariums at the Royal British Columbia Museum and the University of British Columbia. The first known collections in British Columbia for most of the weeds identified in this assessment were between the end of the nineteenth century and soon after the turn of the century. The Sunflower Family: Asteraceae (Compositae) is the largest family of weeds occurring within the South Okanagan. The knapweeds (*Centaurea*) are one of the most well known and ecologically harmful groups in this family; they have caused severe loss of production, environmental quality and aesthetics on thousands of hectares of urban, agriculture, forest and recreation lands. However, new colonies of species such as sulphur cinquefoil, Dalmatian toadflax, leafy spurge and rush skeletonweed are establishing at a phenomenal rate in the Okanagan and many infestations are reaching environmentally severe size and density. The potential distribution of these species is highly significant and must be addressed. Emphasis should be placed on prevention, awareness, early detection and eradication where possible; these are the most practical, economical and effective means of weed management where weeds have not currently been introduced or established.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp

Monitoring Restoration of the Vaseux-Bighorn National Wildlife Area Following Pipeline Construction

Lynne B. Atwood
Genoa Environmental Consulting
Box 20282, Smithers, BC, V0J 3P0, Canada
latwood@bulkley.net

ABSTRACT

A native plant restoration program at the Vaseux-Bighorn National Wildlife Area included 4 treatments: (1) collection and hand broadcast of native herb and shrub seed; (2) salvage and transplant of native perennial grasses, herbs, and shrubs; (3) hydroseeding of native grasses; and (4) application of microbiotic crust. Early monitoring identified successful re-establishment techniques for many native species and a list of herbs and shrubs that should be considered for dryland native seed mixes.

Key words: microbiotic crust, native vegetation, Okanagan, pipeline construction, restoration, Vaseux-Bighorn National Wildlife Area.

BC Gas Utility Ltd. (BC Gas) constructed a natural gas pipeline in the South Okanagan in 1994. The pipeline crossed the Vaseux-Bighorn National Wildlife Area on the west side of Vaseux Lake approximately 5 km south of Okanagan Falls. The South Okanagan is recognized federally as a biodiversity hot spot (Mosquin et al. 1995) and the National Wildlife Area is one of the largest protected areas in the region. The wildlife area is critical habitat for Red- and Blue-listed wildlife and a key component of the South Okanagan Conservation Strategy (Bryan 1996). The native plant restoration program designed for the Vaseux-Bighorn National Wildlife Area represents one of the largest and most complex native plant restoration projects in British Columbia.

This study summarizes the results of a monitoring program carried out in 1996 (Atwood 1996). I discuss 4 components of the program: 1) collection and hand broadcast of native herb and shrub seed; 2) salvage and transplant of native perennial grasses, herbs, and shrubs; 3) hydroseeding of native grasses; and 4) application of microbiotic crust.

METHODS

The wildlife area was divided into 7 geographic areas (GA 1 to GA 7) based on preconstruction vegetation surveys and soil characteristics. Restoration prescriptions were devised for each geographic zone. Construction schedules divided the restoration into 2 stages: GA 6 and GA 7 were revegetated in November 1994 and GA 1 to GA 5 were replanted in March 1995.

Native herb and shrub seed was broadcast into predesignated broadleaf zones if the species was recorded in the local area before construction. In 1996, the effect of broadcast seeding was determined by recording the species in the planted zones. The broadleaf areas were also inventoried for the survival of whole plants, which were salvaged from the right-of-way before construction and replanted in March 1995.

To assess the hydroseeding component of the restoration program, data were collected from 3 1-m^2 quadrats that were systematically located along 18-m transects. These transects crossed the disturbed portion of the right-of-way and were spaced 150 m apart. Data collected included vascular plant species and percent cover, as well as percent cover of litter and bare ground.

The success of seeding microbiotic crust was tested in a split-plot experimental design. There were 2 treatments, with and without microbiotic crust. The crust was applied at 3 different rates (0.55, 0.7, and 0.9 L/m^2), with 3 replications/application rate. Six replications were located in GA 6 (north Plot 1 and 2, seeded in November 1994) and 3 replications were installed in GA 3 (south Plot, seeded in March 1995). Data for percent cover of microbiotic crust and vascular plant species and the number of individuals of each vascular species were collected. Analysis of variance (ANOVA) was used to test for differences between treatment means.

RESULTS

Broadcast Seeding

Over 360 kg of native seed was collected from local antelope-brush (*Purshia tridentata*) shrub-steppe and ponderosa pine (*Pinus ponderosa*) dry forest habitats. The collected

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Table 1. The percent survival of the salvaged plant species and the percent of broadleaf areas with germinants for the broadcast species.

Species	Whole plant salvage (% survival)	Broadcast seeding (% of broadleaf areas with germinants)
Achillea millefolium	not salvaged	94
Amelanchier alnifolia	75.8	1
Antennaria spp.	68.2	11
Apocynum androsaemifolium	not salvaged	5
Arabis holboellii	not salvaged	17
Artemisia campestris	91.7	14
Artemisia dracunculus	73.3	29
Artemisia frigida	97.7	94
Astragalus purshii	25.0	18
Astragalus tenellus	not salvaged	3
Balsamorhiza sagittata	not salvaged	17
Bunchgrasses (5 perennial spp.)	76.0	not seeded
Chrysopsis villosa	60.0	12
Chrysothamnus nauseosus	not salvaged	34
Clematis ligusticifolia	0.0	not seeded
Erigeron corymbosus	not salvaged	51
Eriogonum heracleoides	72.7	8
Eriogonum niveum	90.0	16
Gaillardia aristata	not salvaged	88
Leptodactylon pungens	75.0	not seeded
Lithospermum ruderale	0.0	8
Mahonia aquifolium	42.9	not seeded
Penstemon fruticosus	100.0	not seeded
Phacelia hastata	66.7	8
Phacelia linearis	not salvaged	39
Philadelphus lewisii	100.0	not seeded
Plantago patagonica	not salvaged	49
Purshia tridentata	75.7	not seeded
Rhus glabra	100.0	not seeded
Ribes cereum	100.0	8
Rosa woodsii	79.3	not seeded
Symphoricarpus albus	82.6	not seeded
Zigadenus venenosus	not salvaged	4

shrub and herb seed was broadcast into the designated broadleaf areas in November 1994 and March 1995. Herb and shrub establishment was very selective. Of the 23 species broadcast, only 7 established in >25% of their sown areas (Table 1). Pasture sage (*Artemisia frigida*), yarrow (*Achillea millefolium*), brown-eyed Susan (*Gaillardia aristata*), long-leaved daisy (*Erigeron corymbosus*), and woolly plantain (*Plantago patagonica*) showed no selectivity between geographic areas and may be good candidates for a dryland native plant restoration seed mix.

Little information is available on seed handling techniques for native herbs and shrubs (i.e., collecting, cleaning, storing, stratifying, and germinating). Other broadcast herb and shrub species may have had higher establishment rates if handled differently.

Whole Plant Salvage

During the summer of 1994, 7,600 plants were salvaged from the BC Gas right-of-way in the Vaseux-Bighorn Wildlife Area. The plants were stored over winter and replanted in March 1995. The salvaged species included 12 shrub, 9 herb, and 5 perennial bunchgrasses. Re-establishment success was high. Shrubs averaged 77% survival, bunchgrasses 76%, and herbs 61% (Table 1). Although average survival was high, the timing of the salvage proved critical. Red three-awn (*Aristida longiseta*), a C4 bunchgrass, was actively growing during the salvage and its survival was very low.

Many species responded well to salvage and transplant but not to broadcast seeding. The costs associated with salvage projects could be reduced if species that are difficult to propagate are targeted. For example, saskatoon (*Amelanchier alnifolia*), squaw currant (*Ribes cereum*), and snowberry

Table 2. Five perennial native grasses and 1 annual agronomic grass used in the hydroseed mix.

Grass species used in the hydroseed mix	
Bluebunch wheatgrass (*Elymus spicata*)	Native perennial
Junegrass (*Koeleria macrantha*)	Native perennial
Needle-and-thread grass (*Stipa comata*)	Native perennial
Red three-awn (*Aristida longiseta*)	Native perennial
Sand dropseed (*Sporobolus cryptandrus*)	Native perennial
Annual ryegrass (*Lolium multiflorum*)	Agronomic annual

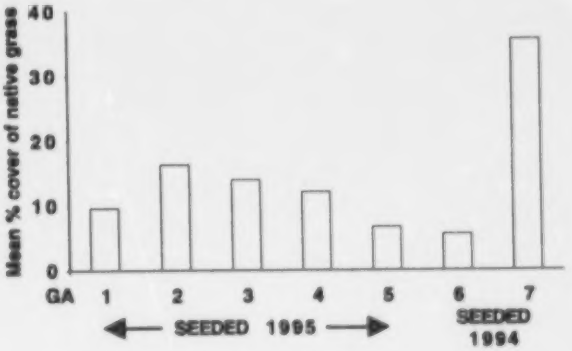

Figure 1. Percent cover of hydroseeded native grasses 1 year after restoration (1996) for the geographic areas seeded in 1994 (GA 6, GA 7) and 1995 (GA 1 to GA 5).

(*Symphoricarpus albus*) are important species for wildlife, but their stratification and germination requirements are difficult and time consuming (Hudson and Carlson 1998, Rose et al. 1998). Salvaging and replanting the shrubs ensures the high wildlife values are returned to the habitat.

Hydroseeding with Native Grasses

The hydroseed mix contained native species that were recorded in the geographic area before construction. Five mixes that differed in species composition and proportions were used and seeding rates varied from 1,747 to 2,078 seeds/m^2. The seed mix contained grasses only, a combination of 5 native perennials and 1 annual agronomic (Table 2). A grass mix was used because broadleaf weeds were a serious problem before construction and grasses would survive any required herbicide applications. In addition to the hydroseed mix, nursery-produced bluebunch wheatgrass plugs were planted throughout the right-of-way.

One year after restoration the percent cover of native grasses and the dominant weed species varied by geographic area. The percent cover of grasses ranged from 5.6% in GA 6 to 35.6% in GA 7; cover in the areas seeded in 1995 fell between the extremes (Fig. 1). The bluebunch wheatgrass plugs established in all areas and were particularly successful in GA 7, which received both bluebunch seed and plugs. Annual ryegrass provided the highest cover of the seeded grasses in GA 2 to GA 5 and GA 7, while sand dropseed dominated the grass cover in GA 1 and GA 6 (Table 3).

Crested wheatgrass (*Agropyron cristatum*) was in GA 7 before construction and was the primary weed in the area 1 year after restoration (16.7%). In all other areas, cheatgrass (*Bromus tectorum*) and diffuse knapweed (*Centaurea diffusa*) were the prevalent weeds. Cheatgrass dominated GA 1 to GA 4 and GA 6 and cover ranged from 6.5% (GA 4) to 31.4% (GA 6). Diffuse knapweed dominated GA 5 (14.6%) but also contributed much of the cover in GA 6 (26.2%) and GA 7 (10.2%).

An extensive weed control program for diffuse knapweed occurred before construction. After restoration, the percent

Table 3. Percent cover of seeded grasses in each of the geographic areas 1 year after restoration.

	% cover of seeded grasses in 1996					
	Aristida longiseta	*Elymus spicata*	*Koeleria macrantha*	*Stipa comata*	*Sporobolus cryptandrus*	*Lolium multiflorum*
GA 1	ns[a]	3.5	ns	2.4	2.6	1.1
GA 2	0.0	2.2	ns	1.8	1.5	10.8
GA 3	ns	5.1	ns	1.0	2.6	5.2
GA 4	1.0	3.4[b]	ns	0.0	0.8	6.9
GA 5	ns	1.3	ns	1.7	1.1	2.7
GA 6	ns	1.1	ns	1.4	2.8	0.2
GA 7	ns	23.3[b]	3.9	1.7	1.0	5.7

[a] ns: not seeded in geographic area.

[b] Geographic area received both seed and plugs. All other areas received plugs only.

cover of the biennial weed was lower in areas seeded in 1995, but higher in areas seeded in 1994 (Fig. 2). The weed control program appeared to reduce competition during the initial vegetation establishment phase. However, the benefits were short lived, likely because the surrounding area was infested with diffuse knapweed.

Cheatgrass was not controlled—more attention should have been paid to this weedy annual. One year after restoration the cover of cheatgrass was equal to or higher than preconstruction surveys (Fig. 3). Cheatgrass germinates early and competes fiercely for soil moisture and nutrients (Martens et al. 1994). Native grass establishment may have been higher if the cheatgrass had been controlled.

Seeding Microbiotic Crust

Fifty-six hundred litres of microbiotic crust were collected from the right-of-way before construction and reapplied with a hydroseeder in 2 areas in November 1994 (north plots 1 and 2) and 1 area in March 1995 (south plot).

Plots inoculated with microbiotic crust propagules averaged 92% higher cover of crust species than plots that did not receive the inoculant. Percent cover ranged from 3.6% in the south plot to 10.7% in north plot 1 and 24.1% in north plot 2. The south plot and north plot 2 also contained a significantly higher cover of vascular plant species in the inoculated plots (south: $F = 19.0$, df = 1, 82, $P = 0.0004$; north plot 2: $F = 4.68$, df = 1, 31, $P = 0.04$).

Three grasses were associated with the inoculated areas in the south plot and north plot 2. Cheatgrass density averaged 129.2 ± 33.1 individuals/m^2 in inoculated plots compared with 20.5 ± 3.2 individuals/m^2 in areas that did not receive microbiotic crust. Needle-and-thread grass (*Stipa comata*), a native perennial, averaged 5.0 ± 0.8 individuals/m^2 compared with 2.9 ± 0.4 individuals/m^2 in non-inoculated areas. Six-weeks fescue (*Vulpia octoflora*), a native annual, averaged 8.0 ± 4.9 individuals/m^2 in inoculated areas and only 0.03 ± 0.03 individuals/m^2 in plots that were not inoculated. The 3 species have similar florets; all are relatively long and slender with hairy or rough lemmas that are awned. The physical characteristics enabled the florets to cling to the mosses in the microbiotic crust.

Microbiotic crust is linked to several ecological functions (Atwood and Krannitz 2000) and its reintroduction expedites colonization. However, the tendency for the crust to store seed immediately benefits habitat restoration, but only if the reintroduced species are native. Species in the microbiotic crust should be identified and weed seeds should be treated before reintroducing the microbiotic crust.

DISCUSSION

The restoration project on the Vaseux-Bighorn National Wildlife Area was a tremendous undertaking because information on the South Okanagan native plant communities was limited and dryland restoration techniques rudimentary. The 1996 monitoring program occurred early in the

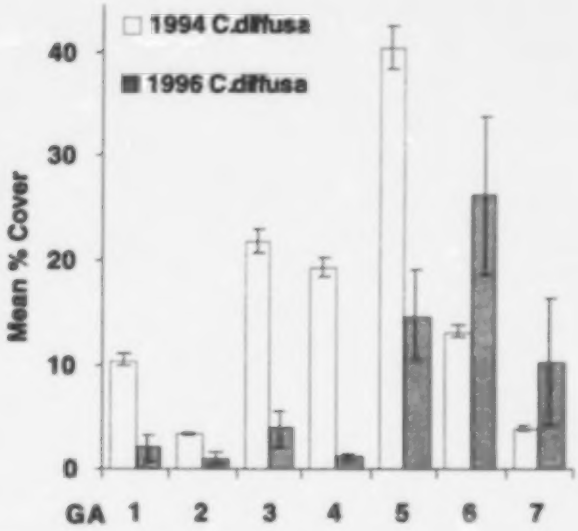

Figure 2. Percent cover of diffuse knapweed (*Centaurea diffusa*) in each geographic area before construction (1994) and 1 year after restoration (1996) (1994 data courtesy of T. Duralia).

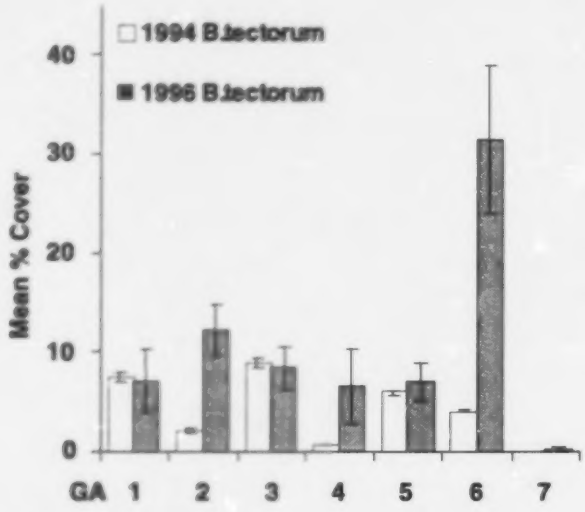

Figure 3. Percent cover of cheatgrass (*Bromus tectorum*) in each geographic area before construction (1994) and 1 year after restoration (1996) (1994 data courtesy of T. Duralia).

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recolonization process; judging the overall success of the restoration project is impossible from this single monitoring event. However, the early monitoring identified successful re-establishment techniques for many native species and a list of herbs and shrubs that should be considered for dryland native seed mixes. The program also highlighted differences in the recolonization ability of native perennial bunchgrasses. This project is the first to document the successful re-establishment of the microbiotic crust.

BC Gas recognized the experimental nature of this project and encouraged the documentation of information and techniques that have subsequently increased ecological restoration knowledge for British Columbia's dryland habitats. Questions raised during the project and the early monitoring program are now being tested in an antelope-brush community outside Osoyoos (Atwood and Osoyoos Desert Society 2000).

The natural habitats of the South Okanagan are disappearing at an alarming rate and those that remain are severely fragmented and extremely vulnerable to weed invasion. Retaining these rare habitats will require restoration, but information on native species phenology, seed biology, and species-specific propagation techniques is lacking. We also lack technical expertise on when to collect seed and what proportions to collect without depleting donor habitats, as well as how best to clean, test, and store the seed. Many of the technical issues could be addressed if provincial standards for native seed were developed. An adequate quantity of high quality native seed is the most limiting factor to habitat restoration today. Without it, re-establishment techniques cannot be tested.

ACKNOWLEDGEMENTS

I thank BC Gas Utility Ltd. for undertaking such a challenging restoration project and funding this monitoring program. Westland Resource Group and Carex Environmental Restoration Group contributed extensively to this project. I also thank T. Duralia (Ph.D. candidate, University of British Columbia) for providing preconstruction vegetation data.

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South Okanagan River Oxbow Restoration

I. Barnett
Interior Field Office, Ducks Unlimited Canada
945-A Laval Cres., Kamloops, BC, V2C 5P5, Canada
i_barnett@ducks.ca

ABSTRACT

The Okanagan River was channelled in the 1950s between Skaha and Osoyoos lakes to facilitate drainage for agriculture and flood protection. This resulted in the significant loss and degradation of habitat in this very unique floodplain as well as an alteration of the natural hydrology of the valley bottom. A number of initiatives have been undertaken over the last 20 years to effect some restoration to the areas adjacent to the Okanagan River. These include water supply systems at the north end of Vaseux Lake and the Haynes Lease at the north end of Osoyoos Lake. Other restoration / conservation efforts were undertaken at Hacks Pond north of Oliver and at an area immediately north of Osoyoos Lake. This paper will describe some of the objectives, challenges and results brought about in undertaking these initiatives.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Habitat Relationships of Endangered Grassland Birds in the South Okanagan

Pam G. Krannitz
Environment Canada, Canadian Wildlife Service
5421 Robertson Road, RR 1 Delta, BC, V4K 3N2, Canada
pam.krannitz@ec.gc.ca

Christoph Rohner
Renewable Resources, University of Alberta
Edmonton, AB, T6G 2H1, Canada

ABSTRACT

According to the Breeding Bird Survey (BBS), all grassland bird species have declined significantly over the last 30 years. Many are either Red- or Blue-listed in British Columbia. We were interested in determining the habitat requirements of grassland birds so that their needs could be considered when developing land management protocols. Livestock grazing has been hypothesized to negatively affect bird populations and few data test this hypothesis, mostly because there are few control sites. We conducted spot mapping surveys of breeding birds in May 1994 and 1995 at 10 study sites that represented a gradient of known grazing history in the semiarid shrub-steppe dominated by antelope-brush in the south Okanagan. We included 3 sites that had not been used by livestock for 75 years. Density of breeding pairs, encounter rates, and the distribution of individual territories across and within the 10 sites were used as indicators of bird response to habitat characteristics. Indirect effects, but no direct effects, of livestock grazing were found. Grazing increased the amount of bare soil at a site, and the distribution of bird territories suggested that Brewer's sparrow and lark sparrow preferred habitats with more bare soil and less grass cover. The inverse was found for western meadowlark while vesper sparrow appeared indifferent to these characteristics. Separate from grazing effects, tree invasion into grassland sites was associated with strong bird community changes. Total species richness and diversity of birds increased with ponderosa pine density, but numbers of grassland bird species were negatively affected. In addition, known nest predators of grassland birds were more abundant in sites with more trees. We concluded that the effects of recent levels of livestock grazing were subtle compared with the effects of tree invasion. If we are interested in maintaining endangered grassland bird populations, tree densities must remain low or be lowered through fire.

Key words: bird diversity, Brewer's sparrow, lark sparrow, species richness, tree invasion, vesper sparrow, western meadowlark.

Though some species are by nature limited in distribution and "rare," most species at risk of extinction do not become endangered or threatened instantly. The Breeding Bird Survey (BBS) provides a wealth of data that has been used to assess trends of bird populations over the last 30 years. A species that is declining overall in number but that still has healthy populations is more easily recovered than a species that has only a few small populations remaining.

The BBS results from the last 30 years show that grassland/shrub-steppe bird species have declined more dramatically, more consistently, and over a wider geographic area than any other group of North American birds in the last quarter century (Saab and Rich 1997). For example, significant annual declines range from 0.6% for western meadowlark (WEME) to 3.7% for Brewer's sparrow (BRSP), with many species in between (lark sparrow, 3.2 [LASP]; clay-coloured sparrow, 1.1; vesper sparrow [VESP], 0.8; grasshopper sparrow, 3.5; horned lark, 1.3). The 3.7% annual decline for BRSP means that there is less than one-third the population that existed in 1965. LASP, BRSP, and grasshopper sparrow are also provincially Red- or Blue-listed though globally significant population declines were not considered upon listing.

STUDY AREA/METHODS

Our research revolves around the understanding of habitat characteristics that affect the breeding success of grassland birds. This study includes grassland bird species that have suffered globally significant declines over the last 30 years, but that may not be provincially listed. With a better understanding of the needs of grassland bird species we hope to influence land use and management decisions so that negative population trends can be reversed.

The effect of livestock grazing on grassland bird populations

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Table 1. Location, elevation, average number of ponderosa pine trees/75 x 250 m section, average percent bunchgrass cover, average percent cover of bare soil, and average linear coverage (m) of antelope-brush, for each of 10 sites in the south Okanagan. Sites are ordered from top to bottom in increasing order of historical grazing pressure.

Site	Latitude	Longitude	Elevation (m a.s.l.)	Size (ha)	No. of ponderosa pine	Bunchgrass cover (%)	Bare soil cover (%)	Bitterbrush linear cover(m, over 120 m)
Site with little or no livestock grazing								
BW	49°13'	119°32'	336	12.5	1.67	45.38	0.57	27.44
KB	49°15'	119°30'	452	12.5	17.0	38.70	13.57	29.02
WA	48°59'	119°25'	355	18.75	0	9.61	28.60	25.77
Sites with some historical livestock grazing								
BO	49°07'	119°30'	352	18.75	0	27.62	12.32	37.58
CWS	49°16'	119°30'	371	10	42.40	20.40	14.42	37.58
KL	49°15'	119°30'	340	25	17.15	6.08	2.03	35.96
OS	49°09'	119°32'	369	12.5	52.33	14.69	19.56	53.05
Site with current and historical and livestock grazing								
WT	49°10'	119°31'	476	22.5	2.67	20.85	24.56	16.97
RG	49°13'	119°13'	339	22.5	2.83	16.10	22.43	18.54
ELO	49°04'	119°29'	356	25	0	25.65	51.69	9.79

has been of interest for many years. It is largely assumed that livestock grazing has a negative influence on population numbers, but few hard data substantiate these assertions. We surveyed breeding birds across 10 similar sites in the south Okanagan that varied in extent of livestock grazing over the past 75 years. The sites were dominated by antelope-brush (*Purshia tridentata*) and were primarily located on the east side of the Okanagan Valley (Table 1). We tried to do this study on sites dominated by sagebrush (*Artemisia frigida*), but did not find any sites that had not been grazed. The 3 sites in this study that experienced little or no livestock grazing in the last 75 years were either topographically isolated (KB), alienated by the irrigation flue that was put in after World War I (BW), or alienated by topography from adjacent grazed lands and kept under private ownership (e.g., ice cream parlour in the 1920s) (WA).

Spot mapping occurred at each of the 10 sites in 1994 and 1995. A 250 m wide belt transect/site was surveyed 4–6 times each year within 3 hours after dawn. Site order was randomized so that sites were not surveyed at the same time each day. The sites varied in length of transect from 400 m to 1 km in length. The location and identity of each bird observed were recorded on a map of the site. The 4–6 map sheets/site were combined to estimate delineation of territories (Bibby et al. 1992). In 1995, determining habitat selection of different bird species was facilitated by describing the vegetative characteristics of adjacent 75 m segments of the belt transect for each site (Table 1). Number of ponderosa pine (*Pinus ponderosa*) for each 75 x 250 m wide segment, and aerial cover of bare soil and native bunchgrasses of 6 randomly placed 20 x 50 cm Daubenmire (1959) plots/segment were determined within each of the 10 sites. Average cover per segment was used when describing vegetation attributes of bird territories, and averages of the segments in a site were used to describe site differences. Nonparametric correlation analysis (Spearman rank) was used to determine significance of associations between species assemblages and habitat variables.

RESULTS

We observed 4,271 birds from 80 different species (Table 2). Thirty-eight of these species were breeding in the 10 sites, and the remainder were transients – either migrants, such as white-crowned sparrow, or visitors from neighbouring habitats, such as yellow-breasted chat (Table 2). WEME was the most commonly observed grassland bird, followed by VESP. We found indirect but no direct effects of livestock grazing. Elsewhere, we have shown that heavier historical grazing pressure increased the amount of bare soil at a site (Krannitz and Atwood in prep.). Here, we found that percent cover of bare soil was one of the habitat variables that influenced selection of breeding territories by grassland birds.

We studied in detail the habitat preferences of BRSP, LASP, VESP, and WEME. Because habitat was measured in 1995, we used only the 1995 survey data for this analysis. On average, BRSP and LASP territories had more bare soil than was found on average across the 10 sites, and less than average grass cover. VESP territories reflected site averages in both variables, whereas WEME territories had on average less bare soil and more bunchgrass cover than was found across the 10 sites. To determine whether these differences were significant, we used randomization tests to determine whether these species were choosing territories based on bare soil or bunchgrass cover, or whether selection of breeding sites was random. The randomization test first considered site

Table 2. Birds observed at 10 sites in the south Okanagan, with the number of times they were observed during all surveys in both 1994, and 1995, and the sites at which they were found in either year. B = breeding, T = transient. Transient birds included migrants as well as visitors from nearby habitats.

Common name	Scientific name	No. of birds	Status	1994 site codes	1995 site codes
Western meadowlark	*Sturnella neglecta*	517	B	BO, BW, KB, KL, ELO, OS, RG, WA, WT	BO, BW, KL, ELO, OS, RG, WA, WT
White-crowned sparrow	*Zonotrichia leucophryx*	434	T	BO, KB, ELO, RG, WA	BO, BW,CWS, KB, KL, ELO, OS, RG, WA, WT
Vesper sparrow	*Poecetes gramineus*	265	B	BO, BW, ELO, OS, WA, WT	BO, BW, ELO, OS, WA, WT
Black-billed magpie	*Pica pica*	226	B	BW, KB, KL, ELO, OS, RG,	BW, CWS, KB, KL, ELO, OS, RG, WA,WT
Rufous-sided towhee	*Pipilo erythrophthalmus*	216	B	BW, CWS, KB, KL, ELO, OS, RG	CWS, KB, KL, ELO, OS, RG, WA
Brown-headed cowbird	*Molothrus ater*	206	B	BO, BW, CWS, KL, ELO, OS, RG, WA, WT	BW, CWS, KB, KL, ELO, OS, RG, WA, WT
California quail	*Calliplepla californica*	195	B	BO, BW, CWS, KB, KL, OS, RG, WA	BO, BW, CWS, KB, KL, ELO, OS, RG, WA, WT
Mourning dove	*Zenaira macroura*	178	B	BO, BW, KB, ELO, RG, WA	BO, BW, CWS, KL, ELO, OS, RG, WA, WT
Chipping sparrow	*Spizella passerina*	165	B	BO, BW, CWS, KB, KL, ELO, OS, RG, WA, WT	BO, KB, KL, ELO, OS, RG, WA, WT
Northern oriole	*Icterus galbula*	159	B	BO, BW, CWS, KB, KL, ELO, OS, RG, WA, WT	BW, CWS, KL, ELO, OS, RG, WA, WT
Western kingbird	*Tyrannus verticalis*	147	B	BO, CWS, KL, ELO, OS, RG, WA, WT	BO, CWS, KB, KL, ELO, OS, RG, WA, WT
American robin	*Turdus migratorius*	130	B	BO, BW, CWS, KB, KL, ELO, OS, RG, WT	BW, CWS, KB, KL,OS, RG, WT
American goldfinch	*Caduelis tristis*	124	B	BO, BW, CWS, KB, KL, ELO, OS, RG, WT	BO, BW, OS, RG, WA
Sagebrush Brewer's sparrow	*Spizella breweri* sp. *breweri*	107	B	ELO, RG, WA, WT	WA
Cassin's finch	*Capsdacus cassini*	103	B	BW, OS, WT	BO, CWS, KB, KL, ELO, OS, RG, WA, WT
Calliope's hummingbird	*Stellula calliope*	90	B	BW, CWS, KB, KL, OS, RG, WA	BW, CWS, KB, KL, OS, RG, WA
Northern flicker	*Colaptes auratus*	81	B	BO, BW, CWS, KB, KL, ELO, OS, RG, WT	BW, CWS, KB, KL, ELO, OS, RG, WA
European starling	*Sturnus vulgaris*	77	B	BO, OS, RG	BO, BW, CWS, KL, RG, WT
Rock dove	*Columbia livia*	69	T	RG	BW, KB, ELO, RG
Lark sparrow	*Chondestes grammacus*	56	B	BO, ELO, WA, WT	BO, ELO, RG, WA, WT
Ring-necked pheasant	*Phasianus colchicus*	49	B	BO, BW	BO, BW, OS, RG
Lazuli bunting	*Passerina amoena*	43	B	BO, BW, KB, KL, OS, WA, WT	BO, KL, ELO, WA

Table 2. Continued.

Common name	Scientific name	No. of birds	Status	1994 site codes	1995 site codes
Pine siskin	*Carduelis pinus*	39	T	BW, OS	
Lewis's woodpecker	*Melanerpes lewis*	36	B	KB, ELO	BO, BW, KB, KL, ELO, OS, RG, WA, WT
Red-winged blackbird	*Agelaius phoeniceus*	32	T	CWS, RG	BW, KL, RG
Brewer's blackbird	*Euphagus cyanocephalus*	32	B	BO, RG	BO, KL, OS, WT
Yellow-headed blackbird	*Xanthocephalus xanthocephalus*	31	T	BO, RG	BO
Mountain chickadee	*Parus gambeli*	30	B		BW, CWS, KB, KL, ELO, OS, RG, WT
Red crossbill	*Loxia curvirostra*	29	T		KL, OS, WA
Rufous hummingbird	*Selasphorus rufus*	29	B	CWS	CWS, KL, RG
Pygmy nuthatch	*Sitta pygmaea*	26	B	KB	CWS, KB, ELO, OS
Clark's nutcracker	*Nucifraga columbiana*	24	B	KB, ELO, WT	CWS, KB, KL, ELO, OS, WT
House wren	*Troglogytes aedon*	23	B	BO, BW, KB, OS, RG, WT	CWS, KB, KL, OS
Savannah sparrow	*Passurculus sandwichensis*	22	T	ELO, WA, WT	BO, BW, KB, RG, WT
Black-capped chickadee	*Parus atricapillus*	19	B	BW, CWS, KL, OS	CWS, KB, RG
Western bluebird	*Sialia mexicana*	19	B	CWS, KB	CWS, KB
Grasshopper sparrow	*Ammodramus savannarum*	18	B	BO, BW, ELO, WT	WA, WT
Killdeer	*Charadrius vociferus*	16	T	RG	BO, CWS, RG
Say's phoebe	*Sayornis saya*	16	B	CWS, ELO	CWS, KB, ELO, RG
Western wood-pewee	*Contopus sordidulus*	15	B	BW, CWS, OS, RG	CWS, KL
Eastern kingbird	*Tyrannus tyrannus*	14	B	BO, BW, OS, RG	KL
Purple finch	*Carpodacus purpureus*	13	T	CWS, WT	
Yellow-rumped warbler	*Dendroica coronata*	13	B		CWS, KL, RG
Canyon wren	*Catherpes mexicanus*	11	T	ELO	KB, KL, ELO, RG
Nashville warbler	*Vermivora ruficapilla*	11	B	KB	KB, RG
Chukar	*Alectoris chukar*	11	B	BO, WT	RG, WT
Golden-crowned sparrow	*Zonotrichia atricapilla*	10	T	BO, KB, WA, WT	
American kestrel	*Falco sparverius*	10	T		RG
House finch	*Carpodacus mexicanus*	9	T		CWS, KL
American crow	*Corvus brachyrhynchos*	9	T		BO, KL, RG
Winter wren	*Troglodytes troglodytes*	9	B	BW, ELO, OS	
Clay-coloured sparrow	*Spizella pallida*	8	T	WA	
Common raven	*Corvus corax*	8	T		BO, KL, OS

Table 2. Continued.

Common name	Scientific name	No. of birds	Status	1994 site codes	1995 site codes
Evening grosbeak	*Coccothraustes vespertinus*	6	T		KB, ELO, WT
Downy woodpecker	*Picoides pubescens*	3	T	KB, OS	
Steller's jay	*Cyanocitta stelleri*	3	T	CWS	KL
Gray catbird	*Dumetella carolinensis*	3	T	OS	
Wilson's warbler	*Wilsonia pusilla*	3	T	KB	WT
Burrowing owl	*Athene cunicularia*	3	B	BO	
Mountain bluebird	*Sialia currucoides*	2	T	CWS, KB	
Orange-crowned warbler	*Vermivora celata*	2	T		KB, OS
Solitary vireo	*Vireo solitarius*	2	T		KB, OS
Dusky flycatcher	*Empidonax oberholseri*	2	T		OS, WA
Western tanager	*Piranga ludoviciana*	2	T	ELO	OS
Rusty blackbird	*Euphagus carolinus*	2	T	RG	
Northern harrier	*Circus cyanus*	2	T	RG	ELO
Long-billed curlew	*Numenius americanus*	2	T	WT	WT
Gray partridge	*Perdix perdix*	2	T	WT	
Rock wren	*Salpinctes obsoletus*	2	T		ELO
Black-chinned hummingbird	*Archilochus alexandri*	1	T	BW, CWS, KL, OS	CWS, KB, RG
Blue jay	*Cyanocitta cristata*	1	T	CWS	
Townsend's solitaire	*Myadestes townsendi*	1	T		KB
Common nighthawk	*Chordeiles minor*	1	T	KL	
Song sparrow	*Melospiza melodia*	1	T	RG	
Yellow-breasted chat	*Icteria virens*	1	T	RG	
Cooper's hawk	*Accipiter cooperi*	1	T		WT
Red-breasted nuthatch	*Sitta canadensis*	1	T		RG
Hairy woodpecker	*Picoides villosus*	1	T		ELO
Red-tailed hawk	*Buteo jamaicensis*	1	T		ELO
Yellow warbler	*Dendroica petechia*	1	T		KL

selection: assuming the birds can choose between all 10 sites, do they select the sites that they breed in based on bare soil or grass cover? Then, within the sites at which a particular species was found, did they select their particular territory based on bare soil or grass cover?

Most of the observed differences were significant. Across all 10 sites, BRSP chose territories with less than average coverage of grass ($\chi^2 = -0.903$, $P = 0.002$), but their apparent preference for bare soil was inconclusive ($\chi^2 = 0.5$, $P = 0.09$). As expected, territories of LASP had significantly more bare soil than average ($\chi^2 = 0.973$, $P = 0.005$), whereas grass cover was similar to the average ($\chi^2 = -0.32$, $P = 0.11$). VESP apparently did not choose sites based on either bare soil ($\chi^2 = -0.071$, $P = 0.71$) or grass cover ($\chi^2 = 0.153$, $P = 0.23$). WEME territories had both less bare soil than average ($\chi^2 = -0.304$, $P = 0.005$), and more grass cover ($\chi^2 = 0.240$, $P = 0.04$). These landscape wide habitat choices in antelope-brush dominated sites mirrored choices for these 4 species in sites dominated by sagebrush (Paczek and Krannitz unpubl. data). When considering only the sites in which each species was found, none of the highly significant differences remained significant. This result suggests that while birds selected sites at which they will breed using habitat variation at a broad landscape scale, territory selection within the chosen sites was less firmly based on habitat variation. More detailed analyses of breeding populations are required to determine the importance of habitat variation in nest site selection with a particular site (Mahony and Krannitz 2000).

The habitat variable most closely associated with bird species richness was density (log-transformed) of ponderosa pine ($r_s = 0.70$, $P = 0.02$, $n = 10$). Bird species diversity (Shannon's diversity index) was even more closely associated with log-transformed tree density ($r_s = 0.78$, $P = 0.0008$, $n = 10$). What's most interesting about these associations is that the relationship is log-normal: there is a threshold number of trees (2–3 trees/75 x 250 m section) that, once reached, results in many more species.

This result should be a good thing: Is not increasing biodiversity the mantra of conservation today? We discovered that this was not necessarily the case, and that the species involved must be carefully considered. For example, what species were favoured with increasing tree density? We separated the bird species into 3 habitat guilds, based on Cannings et al. (1987): forest, ubiquitous, and open habitat guilds. Ubiquitous species included predatory corvids such as black-billed magpie, American crow, and common raven, and the parasitic brown-headed cowbird, while "open" species were those associated with grassland habitats, including listed species and those whose populations were declining significantly. Encounter rate (number/km^2) of both forest and ubiquitous species increased with log-transformed ponderosa pine density ($r_s = 0.83$, $P = 0.002$, forest; and $r_s = 0.68$, $P = 0.03$, ubiquitous, $n = 10$), while encounter rate of grassland, or open species decreased significantly with log-transformed tree density ($r_s = -0.73$, $P = 0.02$, $n = 10$). Therefore, slight increases in ponderosa pine density at a site resulted in negative conservation consequences for grassland birds: not only did the habitat become less suitable for them, but also birds that normally preyed on their nests and young increase in abundance.

MANAGEMENT IMPLICATIONS

The negative implications of tree invasion into grassland sites merit concern and action. Each of the 10 sites surveyed in this study was classified as an AN (antelope-brush/needle-and-thread grass dominated) site (Ministry of Environment, Lands and Parks biophysical mapping), within the Okanagan very dry hot Bunchgrass (BGxh1) biogeoclimatic unit, so that trees were not normally considered dominant. If we are interested in maintaining endangered grassland bird populations, we need to consider introducing prescribed burns to the area so that ponderosa pine density will be reduced. Bird community responses attributable to differences in grazing history were subtle compared with the community changes that occurred because of tree invasion. Grassland bird species responded indirectly to grazing, through changes in habitat, largely bare soil (grass cover was not significantly affected by historical livestock grazing; Krannitz and Atwood in prep.). Different species responded differently and in contrasting ways. To maintain a diverse grassland bird community, we need to maintain shrub-steppe habitat in various seral states. Habitat heterogeneity is the key, and again, re-integration of fire at a small scale would provide a mosaic of habitats suitable for various species.

ACKNOWLEDGEMENTS

We thank landowners for access to the sites: the Osoyoos First Nations Band (ELO, WT, OS, RG, BW); B. Kennedy and G. Kennedy (KB, KL); BC Environment (BO); Puget Properties Inc. (WA); and Canadian Wildlife Service (CWS). We were privileged to speak with A. Thompson, M. Kennedy, and J. Thorp to determine grazing histories of many of the sites. We thank R. Tikkemeijer and M. Gill who performed bird surveys in 1994 and 1995, respectively, and J. Dussereault and R. Handler who collected the vegetation data. We are grateful to K. Prior for giving this talk at the conference. This study was made possible with the generous financial support provided by Habitat Conservation Fund, Vancouver Foundation, Environment Canada, Endangered Species Recovery Fund, and Human Resources Development Canada.

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Reproductive Ecology and Population Viability of British Columbia's Endangered Brewer's Sparrow (*Spizella breweri breweri*)

N. A. Mahony

Centre For Applied Conservation Biology
Faculty of Forestry, University of British Columbia
3rd Floor Forest Sciences Centre
3004-2424 Main Mall, Vancouver BC, V6T 1Z4, Canada
mahony@unixg.ubc.ca

P. G. Krannitz

Environment Canada, Canadian Wildlife Service
5421 Robertson Rd., RR #1 Delta, BC, V4K 3N2, Canada

ABSTRACT

Brewer's Sparrows (*Spizella breweri breweri*) are Red-listed in B.C. due to their restricted breeding range in the southern Okanagan Valley and to their threatened shrubsteppe breeding habitat. North American Breeding Bird Survey data indicate that this species has declined significantly throughout its range, extending through the shrubsteppe regions of the western U.S., by 3.9% per year for the last 30 years. The spatial pattern of this decline indicates that it may be most severe in the core of the range with more stable populations around the range periphery. This patterns shows that British Columbia's population, at the northern range edge, may not only be provincially significant, but, if stable or increasing, could be an important conservation refuge for this species. We are studying the reproductive ecology and population viability of Brewer's sparrows at 4 sites in B.C.'s south Okanagan to determine if these populations are stable and to examine how reproductive success is related to varying habitat quality and food availability. In 1997 and 1998 we banded 101 and 125 adults, along with 26 and 225 fledglings respectively. We also followed nest, hatching, and fledging success for 115 and 258 nests in 1997 and 1998 respectively. We will use Population Viability Analysis (PVA) to investigate which life history stages are key to population dynamics, the rate of population change, and the risk of extinction. The PVA will be refined with another full season in 1999 and a follow-up season in 2000 to survey for birds banded in previous years. Examining the factors that affect reproduction and determining the population dynamics through PVA will allow us to make management recommendations for this provincially endangered and possibly, globally significant population.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999.

Leeks and Marbled Murrelets: Some Thoughts on Endangered Species Education

Briony Penn
Penn and Gunn Associates
224 Meadow Drive, Saltspring Island, BC, V8K 1T9, Canada
penn&gunn@saltspring.com

Donald Gunn
Penn and Gunn Associates
224 Meadow Drive, Saltspring Island, BC, V8K 1T9, Canada

Key words: education.

EXPANDED ABSTRACT

Have you ever noticed that discussions about "the protection of the environment and endangered species" invariably end with the remark that the only solution is to educate the young? I usually feign an attack of chronic indigestion at this point so that I am not forced to be rude. Threatened species and spaces will not wait around for yet another generation. I once had to endure a local politician pontificating at a conference about how nature education kits were going to save our local endangered ecosystem. I had this strong desire to point out that the only possible successful use of those kits was for children to use them as small missile weapons to knock out approaching bulldozer drivers, financiers, and politicians that continued to absolve themselves of any responsibility in mowing down the last few remaining hectares of Garry oak meadows or burrowing owl grasslands.

There is a hard question we have to address and that is: Is educational reform changing fast enough to save species? Take a look at some of our educational institutions today to gauge how they are going to lead our stalwart young down the path to ecological virtue and environmentally ethical behaviour and convince their parents at the same time.

This is a typical horror story. I went to see a new school near Qualicum Beach to see if the planners considered retaining the natural landscape as an example to the young: landscaping and architecture as pedagogy. I approached the school with anticipation. The area as I knew it had been classic coastal Douglas-fir forest, with maples and arbutus circling a hill that was covered with oak. I envisioned a school nestled into the trees with dense thickets of salal to create forts and tunnels for play and solitude. I imagined big trees and old snags under which teachers would lead discussions as to how woodpeckers create nest cavities for squirrels or how gravity was first conceived or what volume of water a tree can absorb. There would be the pond to do water life studies and old nurse logs to sit upon and recount aboriginal stories of life hundreds of years ago. The values of indigenous species like yew to cancer research would be pointed out in the forest as students explored their surroundings, each other, and learnt the importance of biological diversity.

I have been known to suffer delusions so it is probably not a surprise that the scenario that greeted my eyes was rather contrary to expectations. Sir Isaac Newton, for example, would not have theorized on the law of gravity at this school, for there is nothing within a radius of 3 hectares that could even be loosely described as living vegetation, let alone a tree. It would take a singular act of vision to theorize on any physical law, action or principle in these grounds, as there is nothing there but a peculiar green turf. There are no secret places to go. There are no places to tell stories. There are no plants, insects, animals, or any trace of what came before to encourage a contemplation of history. The tedium of the site stretched before me and I felt like committing great acts of juvenile delinquency like hurling education kits at the perpetrators of this massacre. I was told by an official that bushes and trees caused students to commit acts of indecency and illegality so they removed them all hoping that they might commit these acts in the malls instead.

The good stories come from places like Sundance, an alternative school in Victoria. The kids, the parents, and the teachers struggle like all others to juggle the demands of this complicated world and in 1 small way they have achieved a victory. Pushing up amongst the blades of grass under a canopy of oaks are the buds of a patch of native wildflowers, collected and grown by the students from a site that was bulldozed. The patch is surrounded by a little fence constructed by the kids woven in and out with wools and ribbons, behind a patch of snowberries where they have their fort and conduct childhood experiments out of the eye of the adults. The industrial lawn-mowers have agreed to halt their mowing in this patch for the spring, and the neighbourhood has joined in the anticipation of watching them bloom.

This all happened because some parents and the teachers felt their children have enough contradictions in their lives without being preached at about conservation with biological deserts for schoolgrounds. They recognized that without the raw material to experience, no number of nature education

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kits are going to create a love of the natural world. The solution lies in a little judicious reeducation of ourselves first.

Trying to accomplish even these small measures though is monumental in effort. Any kind of teaching about endangered species is a little like being a rider of the apocalypse. When you ride into town, the writing is on the wall. Your little message is part of that last nail in the coffin for a way of life and the natural wealth of a place, the two spiralling into oblivion together. You are called in when there is nothing much left but artifacts and aspirations for a new industry in restoration. And you are not just educating about restoration of natural systems and species. You are really there to teach about restoration of community rifts, caused as economies fall apart; restoration of the historical reality of the era not just the nostalgia; and then restoration of the land that was cut down or dug up to fuel that community.

I have been at the death of many communities in Britain and Canada: fishing, logging, mining, farming, wool manufacturing, rural, and aboriginal communities. I've written educational material on the demise of species from mammoths to murrelets, the end of ecosystems from the pine forests of Scotland to the oak meadows of Victoria, and the close of cultures from Celts to Cowichan fishers – diminishing diversity in every realm of life. The most successful thing about any of these projects is a little like what Desmond Tutu is doing with his Truth and Reconciliation Committee – a place to get it out on the table then put it behind you and hopefully maintain a broad respect – a process. The impact on maintaining the species is more difficult to gauge.

The first project that I began to understand the power of the process was in a mining museum in the north of England, called Woodhorn Colliery Museum. I was assigned to it 12 years ago as a junior designer. This is a region that has lost more species than it will ever remember. And it was all land that had once looked like Canada and probably enjoyed such Celtic place names as Valley of the Grizzly, Hill of the Wolf. It has suffered loss of species since the Bronze Age through human destruction of habitat and overhunting. The mining industry over the last 200 years finished it off. Slag heaps rose where valleys had once been, forests were levelled, mine shafts ran 3 miles out under a sea that had run out of fish, and the smoke turned the sky black, until the coal ran out.

Fresh from Canada, the questions you ask when you get to these places are: Is protection of endangered species a new idea? Did anyone care in the past? If they did, what did they do that didn't work? Do communities just exhaust the earth then move on? Is there any role for education? Fresh from Canada you also assume they might be happy to lock up all the grimy tools as artifacts in the museum and get to the real business of covering the slag heaps with earth, detoxifying the tailing ponds, planting the forests, and reinstate the wildlife and Celtic mythology and songs in a couple of Valleys of the Bear. What I found was an overwhelming need from these people to say goodbye to leeks, racing pigeons, and wash tubs.

I spent 2 years on that job, on and off, eating "mushy peas" and listening to stories about champion leeks and social clubs and drilling at the face and pithead baths. By the time it was finished, the museum consisted of reconstructions of all these aspects of mining life, as well as the record of child labour, mining accidents, the degradation of the land, and an art gallery of paintings done by the pitmen painters of real life down the mine. With the help of an historian/curator who collected the archival material and wrote the interpretive text, the miners put the museum together with us. In the course of doing a reconstruction of a mine I learned to understand the Geordie accent and earned a rank somewhere above dustcart boy, but something of the gist of the Valley of the Bears and Celtic sensibilities crept into the conversation and the exhibit.

Woodhorn Colliery Museum was opened on a warm spring day by the leader of the British Labour Party. Every person in that community came to it. They walked to the museum by a path that went by a restored lake and over a newly forested hill down to the old restored pit head buildings that housed the winding gear, the museum, and their old lives. They were lined up in a big long queue that curved around the museum and the old social club brass bands sparkled in the sun. The morris dancers danced and the calliopes steamed their tunes away for the kids. The old women cried when they saw the reconstructed front rooms of their cottages. One of the miners who had been a supervisor in his day, and had naturally taken over the job of ordering me around, gave me a gift of a photograph he had taken. It was of an old statue of a miner that had been erected to honour the dead of countless mining accidents. His picture had captured a moment when a real dove had landed on the miner's stone hand. He knew I would like it.

Today in British Columbia, as primary industries collapse and ecosystems disappear, resource communities here are doing their grieving, and environmental education is gathering steam. The role education can play in raising awareness, community development, eco-tourism potential, and creating a mindset for restoration projects is largely dependent on how well we say good-bye to the past and how much we include the present. I have used 2 examples of small visitor centre projects completed in the last few years in areas where concern over endangered species and spaces was the primary reason for these exhibits. They embody a lot of the ideas that might be used at any level of education for endangered species and on which I found the British experience was a useful one to draw: Gwaii Haanas/Queen Charlotte National Park Reserve for Parks Canada and the Rainforest Interpretive Centre for the Long Beach Model Forest in Tofino.

In both projects, some of the essence of Woodhorn was adapted. Both these British Columbia communities are

shifting rapidly from resource extraction to eco-tourism and are highly polarized. Types of humour, and skills don't transfer from 1 way of life to the other very well. The introductory remark when we started on 1 of the projects was, "there are 3 things we can't mention because of the politics: trees, fish, and Indians." They might well have added other controversial things such as marbled murrelets.

With the 2 visitor centres and a limited remit, we focused on using local skills, craftsmanship, and materials, and tried to look for ways of engaging the future with the past. The highlight of the Gwaii Haanas centre was a 50-metre-long fabric frieze that depicted the natural and cultural history–marbled murrelets and Keen's long-eared myotis to steam donkeys and miner's picks – put together by a team of local women from all politics and cultures. Appliquéd Haida Steller's sea lions swam amongst knitted octopus, while embroidered eighteenth century sea otter ships bobbed on the batiked sea in front of woven mining communities. An exhibit on tides was built of cranks and handles off fishing boats by a fisherman, with a piling adorned by intertidal life sculptures done by another team of local sculptors. A bathymetric model of the island was done by an islander who knew the coastline. As many elements of the museum as possible were designed for and created by the skills that existed in the community. We fit as many endangered species on as we could.

The same principles were applied for the Tofino Rainforest Interpretive Centre. A rainforest mural was completed by 6 local artists under the direction of a master muralist. A tile mosaic was done by 1 of the outlying village schools to interpret the source of native clan symbols, and a series of quilts were done by another local group to interpret the biological values of old-growth forests. A doll's house was built to interpret the use of timber in a household, another woodcraftsman created a wooden globe to show the distribution of the world's rain forests. All these activities inevitably reach out to a wide number of families, and channels for communication and exchanging ideas are opened.

Both centre openings were happy affairs, lots of music and different members of the community full of talk of trees, fish, and Indians. I threw in some anecdotes about leeks and racing pigeons and maybe a statue for the loggers who had lost their lives in those Valleys of the Bears. Whether either of these projects will help protect endangered species I'll never know, but at least both communities can associate something other than guilt and economic ruin with endangered species. Who knows, maybe 1 day a marbled murrelet might alight on the statue's outstretched arm.

In summary:

1. It can't get as bad as Britain (we can't let it).
2. Honour what was good about the past first before you talk about the future.
3. Be honest about what is threatening endangered species and put it in the light of day.
4. There have been people trying to save endangered species forever. Their voice is rarely recorded. Let their memory sustain you.
5. Local place is everything.
6. Integrate environment education into every facet of life from architecture to nursery rhymes.
7. Integrate endangered species into the economy however you can.
8. Don't lose your sense of humour.
9. Embrace the paradox.

Conservation Tools and Process

Jan Garnett[1]
The Nature Conservancy of Canada
404-3960 Quadra Street, Victoria, BC, V8X 4A3, Canada
jang@natureconservancy.ca

Key words: conservation covenants, non-government organizations.

SUMMARY

Temporary Conservation Measures

1. Stewardship agreement.
2. Licence.
3. Lease.

Permanent Conservation Measures

1. Conservation covenant registered on the title of private land, with thorough baseline documentation and regular, well-documented monitoring. This covenant is binding on all future owners of that property in perpetuity.
2. Bequest or donation of land, with secure conservation ownership and/or a conservation covenant registered on title.
3. Purchase acquisition, followed by secure conservation ownership and/or a conservation covenant registered on the title.

Evaluation, Selection, And Completion

Summary of the steps a non-government conservation organization takes toward project evaluation, selection, and completion:

1. Identify the most ecologically valuable and endangered or threatened ecosystems and areas of the region in which the organization works. Have priority ecosystems or heritage values established up front. Proactively pursue conservation opportunities within those parameters, without waiting for outside requests. Funding must be secured for even this initial site selection process. A master strategic conservation plan on an ecoregional scale is ideal for a large-scale conservation organization like The Nature Conservancy of Canada (NCC) or a provincial government agency.
2. Screen incoming project requests by asking for as much information as possible to be supplied by the proponent before a potential site visit. Evaluate these requests by other conservation groups, government, non-government organizations (NGOs), individuals, or landowners based on established scientific and feasibility criteria. A huge percentage of requests for acquisition help have to be refused or referred to a regional conservation organization. A higher percentage of conservation covenant possibilities survive the process, depending on available funding, location in province, strategic factors, and staff time.
3. For projects identified as possibilities through this initial screening process, a site visit by the NCC Projects Coordinator (or equivalent in other organizations) and scientific advisor(s) is arranged with the landowner. If there are several sites to be considered in 1 area of the province the visits are usually done during a single extended field trip for cost reasons. This is the same for conservation covenant properties. Most evaluations are obviously best done from spring to autumn, and arranged at the convenience of the landowner and other people involved. At any point, specific expertise may be necessary for evaluation or in negotiations.

Priority Acquisition Projects

For the very few acquisition projects that rise to the top each year as being the most endangered, the best ecologically, and/or the most "do-able" from a timing and funding point of view, these steps are followed:

1. An independent appraisal of the value of the property is obtained if one hasn't already been done.
2. All the information about each property is prepared for presentation to both our Regional Board of Directors and the National Executive Committee, where projects compete with each other for acceptance by these volunteer boards. If an offer is presented to the landowner, it is always subject to acceptance by both these boards, as well as availability of funding. Typically, an offer is littered with terms and conditions, to make sure that every concern is addressed before the deal becomes unconditional, at which point we are legally committed to purchase the property.
3. When the Agreement of Purchase and Sale is finally agreed (very few NGOs can justify paying more than fair market value for a property) the fundraising strategy is finalized, conservation partners are identified, the media campaign is begun, and fundraising is begun in earnest. From the beginning of negotiations with the owner to when the property is actually acquired can be a very long

[1] Present address: The Nature Conservancy of Canada, Suite 202 - 26 Bastion Square, Victoria, BC, V8W 1H9

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

and laborious procedure. That makes it even more important to identify the most important and feasible properties that will stand the test of public scrutiny, potential donors' scepticism, and competition for funding.

4. Occasionally, there is a "major donor" who will want to play a key role in protecting a specific habitat, species, or area of the province. Then the process is more one of finding the right match and negotiating the purchase, with less emphasis on fundraising.
5. Right at the beginning of an acquisition project there is a clear need to know what the eventual ownership, designation, and management of the property will be. In the case of NCC, transfer of title to another organization or a government agency is almost always accompanied by the placing of a conservation covenant on the title, to be held by NCC. This eliminates the possibility that at some point in the future the property might be used for purposes contrary to the conservation and enhancement of its natural values. In those cases where we plan to continue holding title to the property we identify a suitable management agency and work out a management plan.

Good Candidates For Conservation Covenants

The steps for projects identified as good candidates for conservation covenants are:

1. The landowner contact program begins in earnest, and can last through many months (sometimes a year or more). General information about covenants is given to the landowner(s). A covenant template document is given to them and explained, and questions answered. Site visits are done with scientific assessment personnel present on 1 or 2 of the visits to identify and inventory the natural features of the site.
2. Photographs and videotapes are taken, and baseline documentation of all this information is compiled so that the covenant is legally enforceable and can be monitored properly.
3. The landowner's family, lawyer, and, often, accountant are brought into the process to ensure independent advice (e.g., tax implications) and full disclosure.
4. Negotiations on the terminology and content of the covenant continue, and any necessary professional survey and/or appraisal work is done.
5. If everything goes well, the covenant is finalized, signed by the landowner(s) and the conservation organization(s) (or government) who will be holding the covenant, and the document is registered on title, along with the baseline report, survey etc., at the Land Titles Office in Victoria.

Local support is often one of the most important indicators for eventual fundraising success. As with any charity, volunteers at every level are extremely valuable for a conservation non-profit organization throughout the process, and continue to be so after project completion, when the property still needs monitoring and stewardship.

Everyone involved in conservation work should remember to celebrate the successful completion of a project with everyone who was a part of it or helped. Besides the need for those who have worked hard to celebrate the success with the local community, the media attention can help with fundraising for the next project, and with establishing credibility with landowners who may be considering donations or covenants.

Habitat Acquisition Trust: Creating a Conservation Legacy

A. MacDonald

Habitat Acquisition Trust, Land Stewardship Office
P.O. Box 8552, Victoria, BC, V8W 3S2, Canada
HAT@islandnet.com

ABSTRACT

The Habitat Acquisition Trust was established in 1996 to protect locally significant habitat, endangered spaces and the species at risk they support on southern Vancouver Island and the Gulf Islands. HAT accomplishes that goal with three conservation tools. Acquisition projects like Ayum Creek in Sooke, British Columbia, helped to protect an urban salmon stream and provided one of the few protected areas for nesting purple martins in British Columbia. Conservation covenants with individual landowners and different levels of government allow HAT to protect habitat without purchasing it. Finally, public education through initiatives like the Habitat Acquisition Trust Land Stewardship Office are creating new bridges between scientists, conservation organizations, landowners, politicians, decision makers and the public. The HAT LSO was established in September of 1998 with seed money from the Capital Regional District and the Provincial Capital Commission. Its mandate is to provide a "one-stop-shop" for conservation information and to help facilitate greater levels of efficiency, cooperation and partnerships for those working for the conservation of endangered spaces and species. This talk will outline the road to the present and look at our future goals.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

When is Too Little Enough? A Successful Eleventh-Hour Rescue of Vancouver Island's Northernmost Garry Oak Habitat

George Sirk
Box 83, Whaletown, BC, V0P 1Z0, Canada

SUMMARY

Only 1% of British Columbia's original Garry oak habitat now remains, and many of those trees are considered "the living dead"—they have no offspring under them, just lawns and tulips or big, fat cows.

The subspecies of the large marble butterfly, *Euchloe ausonides*, is now extinct. It enjoyed Garry oak habitat during its presence on Earth. It's toast. Nine subspecies localized to this habitat are now considered extinct, endangered, vulnerable, or just plain "up the creek." Why? Well, with only 1% of the original oaks remaining (if you don't count all the pet ones, like those in Victoria that majestically grace a lawn, or the perfect mushroom-shaped beauties that dot the cow fields like sentries), the wild stock can be measured in a fraction of a percent.

Here is a little vignette that speaks volumes for the Garry oaks of bygone days. In Courtenay, in 1997, as the new Regional Director for Cortes Island, I stumbled into a plan to build an ice arena and put its parking lot on the northernmost wild copse of Garry oaks in British Columbia. It had been condoned in a 1996 public referendum. The bulldozers were being unloaded and the architects were in a Titanic frenzy to complete the detailed drawings at about the time I entered the picture, on a routine tour of public facilities. My jaw dropped when I was shown the nearly 1-hectare Garry oak forest, consisting of about 60 mature trees and hundreds of seedlings, seemingly destined to be levelled to make way for progress! in the name of a parking lot! Oh, Joni Mitchell, you know how I feel: "They paved paradise to put up a parking lot."

How do we explain this oversight? The regional district did not do an environmental assessment of the site—it was not required! The natural history societies thought the ice arena was going somewhere else and the public was blissfully ignorant that the trees were doomed. The Ministry of Environment Data Conservation Centre had no record of the grove because their aerial pictures only show copses of more than 1 hectare. ...It had fallen between the cracks, too little to be enough!

Convincing the 19-member Comox-Strathcona Regional District Board to put the parking lot somewhere else was a long, arduous process, but I was determined to drive home the importance of this grove. I tossed in a bit of public insurrection, with naturalists Allan and Betty Brooks leading packs of tree-huggers, and the associated media feeding frenzy—all storming the board room looking for sanity. Meanwhile, I pictured myself chained to the uppermost limbs of the 400-year-old, great-great-great-grandparent oak, thus throwing politics back into the annals of our arboreal past (where it flourishes today) and where it surely belongs anyway.

I hadn't just tipped over the apple cart—I'd taken all the wheels off. To hear me speak you'd think that I was a closet botanist with chlorophyll for blood and a family that led back to ancient oaks, not to Australopithecus at all!

In the meantime, Dr. Adolf Ceska, ecologist with the Conservation Data Centre, prepared an appraisal in support of the grove. He underscored the grove's rarity: "the species composition of this stand is so unique that it is difficult to compare it with any other stand in British Columbia." Well, you know how fussy botanists can be. Toss in a bit of snowberry and cascara and a big dose of wild ginger and a white fawn lily here and there, and they make it sound like we're witness to genesis in action! Thus, when it appeared the board was unwilling to follow his recommendations, Adolf surmised that: "the board seemed determined to destroy this extremely unique stand of Garry Oak."

The board was dumbfounded; they would be guilty of destroying a one-of-a-kind jewel in Courtenay's crown. So the architects were put to work to reconfigure, redesign, and reposition the arena and the parking lot. But, alas, only the parking lot was relocated—the ice arena "had to stay," thereby sacrificing half the habitat and 8 200-year-old trees.

It is extremely lamentable that the "critical size," which was already in question, was now a depressing half-hectare. But more than 50 trees (200 years old) and a few oldies survived this blunder. The best part is that the oaks produced acorns in 1998, which have sprouted and will join the hundreds of saplings in this microhabitat.

For me, the warmest reassurance comes of knowing that the grove should outlive the arena; if we can get it out of the

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current lease contract and into a covenant that will protect it in perpetuity.

The obvious way of dealing with this dilemma is to protect the trees outright, through legislation. I discovered that under the Municipal Act a municipality may protect a tree species for its rarity or beauty, but a regional district may not. Why is this? A regional district has a modicum of general protection for trees built into Bill 26 and Official Community Plan and Covenants. However, I suggest to all that the future for a particular species, be it bird or orchid or tree, needs to be protected by a "stand-alone" provincial Endangered Species Act.

The majority of British Columbians know that the dogwood tree is protected, and not because of the penalties associated with its destruction—a paltry $25.00! The awareness alone that the dogwood is special encourages governments and public to treasure the tree and plan developments and habitat protection accordingly.

In September 1998, I brought a motion (B69) forward to the Union of British Columbia Municipalities Convention floor, which recommended that the Garry oak have the same status and practice of being protected as the dogwood. B69 was endorsed as a recommendation to the provincial government. Sadly, to date, they have chosen to ignore it.

Since the federal Endangered Species Act is nonexistent, it is as good for life on Mars as it is here. And the British Columbia effort is pathetic—Red list, Blue list, let's call it listless! Moving the deck chairs around on the Titanic just isn't good enough.

A senior Ministry of Environment official informs me that the place Garry oaks belong is in regulations, not legislation. This is a dangerous, bizarre concept. It means survival of a species is in the same category as the length of a septic field or the number of toilets a bar requires.

Shall we leave all our species—oaks, arthropods, tiger salamanders—to chance? Chance got us all here, but is chance going to get us out?

What about the less than 1% of Garry oaks that are left? Is there little enough left that the Garry oak needs protection now?!!

When is too little—ENOUGH!

The Conflict Between Urban Development and Species and Habitats at Risk: Can There be a Resolution?

T.T. McIntosh
Golder Associates Ltd.,
202-2790 Gladwin Rd., Abbotsford, BC, V2T 4S8, Canada
tmcintosh@golder.com

ABSTRACT

As the human population increases in British Columbia, so does the development of homes, industries, and associated infrastructure. Concurrent with this, greater real and potential impacts are being observed within or adjacent to habitats and to individual species at risk. Most of the ecological awareness regarding habitats and species at risk is either government-driven (e.g., by the Conservation Data Center, Victoria) or public-driven, especially at the local level (e.g., naturalist groups, trained individuals). At the interface of urban development and ecological awareness, however, there is often conflict, mainly based on the perceived polarity between economic interests and ecological values. Although this conflict is not new, the understanding and management of such conflicts is increasing. There are few, if any, formats available that are designed to reduce potential conflict. Recent provincial legislation intended to help resolve conflicts and guide future development includes the Fish Protection Act and the Farm Practices (Right-to Farm) Act. At present, federal and provincial legislation is being investigated that is perceived as a mechanism that will drive development in a more "ecologically-sustainable" direction. Various municipalities have developed policies and by-laws that must be followed when development is undertaken, yet there are still questions about their practicality and effectiveness. A number of possible solutions to resolving conflicts may include more protective legislation, education, and awareness training. Is there a potential, based on education, stewardship, and economic give-and-take, to reduce conflicts without the aid of government policies? These issues will be explored and discussed within the framework of some examples from the lower mainland.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Why Whoop It Up About Whoopers?

Nahanni Ackroyd
Interpretive Programs, Calgary Zoo
Calgary, AB, T2M 4R8, Canada
education@calgaryzoo.ab.ca

Key words: breeding programs, Calgary, interpretive programs, whooping crane, zoo.

In 1998, the Calgary Zoo opened a brand new exhibit, the Northern Forest, as part of our Canadian Wilds area. Animals on display in the Northern Forest include moose, caribou, river otter, beaver, barred owl, lynx, lots of insects, and of course, whooping cranes.

The zoo has been part of a captive breeding program for whoopers for several years, and some of the captive birds have not yet selected mates, or cannot to fly properly, so they will probably never be able to breed. It was from these birds that our 2 whoopers were selected for display. They turned out to be 2 females, called Borden and Cleopatra.

With the whooping crane exhibit was a new outdoor amphitheatre that looked into the whooper enclosure, and beyond that, into the moose area. For interpreters, the challenges faced included

- how to get visitors to the new area (which is geographically away from the core of the zoo);
- how to get visitors excited about birds (typically our program areas centre on megafauna like elephants, tigers, and gorillas, although we are moving toward more ecosystem-focused programs);
- how to bring visitors up to date on whoopers (many older visitors thought they remembered hearing that whoopers were extinct, and many visitors just walked by the "big white birds"); and
- how to interpret a new area—there are no previous programs to borrow from, no props already built, and interpreters already carry out programs in 7 other areas.

Some of these issues could be answered in the program content, while others were addressed in the style of program, and the on-site advertising. Others, like building props, had to wait until the program was thought out, and just took time.

The zoo as an organization drew visitors to the new exhibit through off- and on-site advertising, and by contracting an outside theatre/interpretation company to present a Northern Forest themed program on weekends, when there tend to be more visitors to the zoo. The contractors performed a 2-person musical program that lasted about 30 minutes. Typical zoo programs last 20 minutes, as we are continually upstaged by live animals. As the summer progressed the contractors and the zoo interpreters saw a rise in audience numbers at the new amphitheatre. Word of mouth was on our side, and I expect the site will be even more popular this summer.

How did we get people excited about big birds? For us, the answer was an irreverent 2-person program. Due to scheduling, we can rarely present 2-person programs because each interpreter already does 3 individual programs a day. However, we wrote a program that was fun to present, so that some interpreters were willing to take on a fourth program to help out. Unfortunately, the whooper amphitheatre is in the hot sun, and our program was scheduled for 1430 every day. To give the audience a rest while still participating, we had the audience actively participate in the introduction; after that, all they had to do was sit back, laugh, learn, and vote on bachelors. We decided that the Dating Game format would provide lots of opportunity for humour and information dissemination.

Because the interpreters were having so much fun with the program, it was easy to promote the program at earlier, traditionally more popular programs. Our 1300 "Trunk Tales" draws between 300 and 500 people, a perfect time to promote "Whoop It Up," with lots of time for visitors to walk out to the site. Who doesn't want to vote on likely and unlikely bachelors?

Once we got people to the new amphitheatre, what did we do? The goals of the 20-minute program were

- to introduce the Northern Forest, not as a blanket of trees, but rather, a patchwork quilt of deciduous and coniferous trees, openings from forest fires, rivers, and lakes, and if you look really closely in this marsh, you might just see...Wanda Whooper;
- to introduce some of the creatures who live in the Northern Forest as possible dates for Wanda: a caribou, a great grey owl, a dragonfly, and, finally, the unheard-of, a fourth bachelor—a male whooper;
- to illustrate the critically low numbers of whooping cranes through Wanda's desperation to find a mate; and
- to have a lot of fun, both audience and interpreters.

This format worked well for us, and I expect the program will grow in coming years.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Protected Areas Education: A Pilot Project of B.C. Parks and Schools in the Garibaldi/Sunshine Coast Region

D. G. Blair-Whitehead
Whitehead Environmental Consultants Ltd.
RR #1 CH-11, Bowen Island, BC, V0N 1G0, Canada
dghwh@ibm.net

Key words: education, Howe Sound, protected areas, provincial parks, Sunshine Coast.

British Columbia (BC) Parks, Garibaldi/Sunshine Coast District, is sponsoring a protected areas education project. The goal of the project is to introduce and facilitate the use of the environmental education guide *Protected Areas: Preserving Our Future* (P.A. guide) (Staniforth et al. 1996) in the 2 school districts, Howe Sound/Whistler and Sunshine Coast, within their area. The 138-page guide was published with sponsorship from the British Columbia Ministry of Environment, Lands and Parks and Ministry of Forests, and Canadian Heritage Parks Canada. The guide explores the concepts, values, and practices that underlie protected areas by considering the "whats, whys, and hows" of protection of things we value through student activities and teacher background information.

The project objectives include completing a pilot project with 4 teachers from Valleycliffe Elementary School in Squamish and delivering professional development workshops to district teachers during the 1997/98 and 1998/99 school years. The final objective is to have intermediate-level (grades 4–7) teachers throughout the Garibaldi/Sunshine Coast District use the P.A. guide without ongoing support and funding from BC Parks.

The pilot project (September 1997–March 1998) provided support and training on the P.A. guide to a select group of teachers (grades 3–7) at Valleycliffe Elementary. The teacher training and support was done in various ways based on an assessment of teacher requirements. The contractor adopted an individualized, flexible approach to teacher training.

Teachers clearly felt they needed training and support in the areas of habitat/field studies as well as locally relevant case study material. Especially important to them was support for applying local situations (habitat/field studies) to science Integrated Resource Package (IRP) Learning Outcomes. Thus, 4 of the 6 training sessions were spent on habitat/field studies. The remainder of the training was on the Mapping for Decision activity (Chilcotin and Alice Lake case studies).

One of the outcomes of the pilot project was the development of a supplemental document to the P.A. guide focused on Alice Lake Provincial Park. The *Alice Lake Case Study* (Blair-Whitehead 1996) is intended to complement the P.A. guide and presents background information and activities for a local protected area that is familiar to many students. The *Alice Lake Case Study* was used in 2 pilot classrooms and was presented to teachers throughout both the Howe Sound and Sunshine Coast School Districts via professional development workshops during April 1998. An additional case study on Sargeant Bay Provincial Park, located on the Sunshine Coast, is currently under development. This case study is modelled after the *Alice Lake Case Study* and was introduced to teachers at a recent professional development workshop in Sechelt, B.C. Both case studies will be available to teachers from the BC Parks district office.

The BC Parks' contractor and Teacher Professional Development (Pro-D) committees worked together to introduce teachers to the P.A. guide and local case studies. Pro-D committees meet throughout the school year and coordinate teacher workshops and conferences at provincial or district Pro-D days. Contact was made with the Pro-D Coordinator early in the school year for scheduling protected areas workshops in district schools. Three workshops have been held to date with 27 teachers participating. Workshops focus on introducing the background information in the P.A. guide, hands-on activities, and discussion of "best practice" for implementation of a protected areas unit within classrooms. All participants receive a P.A. guide and a local case study to use at their own school. Further communication with teachers has been through periodic newsletters highlighting protected areas, upcoming workshops, and materials.

CONCLUSION

Regular contact with Garibaldi/Sunshine Coast teachers through Pro-D, newsletters, and workshops has been fundamental to the success of this project. Many teachers are enthusiastic about implementing studies on protected areas with their students. Teachers of grades 3–12 indicate that the

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P.A. activities meet many of their IRP requirements. As well, they feel the local aspect of the case studies is vital to the success of any P.A. study. BC Parks will continue make these case studies available to teachers and work to increase integration of their own programs with classroom curriculum.

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What Do Siberian Tigers and Rocky Mountain Grizzlies Have in Common?

Nahanni Ackroyd
Interpretive Programs, Calgary Zoo
Calgary, AB, T2M 4R8, Canada
education@calgaryzoo.ab.ca

Key words: Calgary, interpretive programs, zoo.

Lots, if you're trying to present global conservation messages in Calgary! The Calgary Zoo has taken on the many roles of the new zoo, from international and national conservation initiatives to educational leadership.

The Calgary Zoo has a large education department, with 11 full-time staff, and over 20 seasonal and part-time staff. The department has expanded from a staff of 3. The education department now has staff who work in particular areas, such as horticulture programs, value-added programs, revenue-generation programs, Zooschool, docent coordinators, and audiovisual media specialists.

I work in the value-added section, coordinating the summer interpretive season. In 1998, we presented programs in 8 areas around the zoo. Because our collection represents animals and plants from around the world, so does our programming. Program areas this last summer included Siberian tigers, Asian elephants, the African Bushveld, Rocky Mountains, boreal forest, gorillas, harbour seals, and dinosaurs. All seemingly diverse topics.

One of the overall themes for the Calgary Zoo, as a new zoo, is conservation. We really don't collect animals simply to display them anymore. In fact, most of our animals were born in captivity. We try to get the most educational value out of a particular animal's captive life. In our interpretive programs we aim to entice the visitor to learn more about that species, and about the threats to that animal's survival in the wild.

In 1998, over 140,000 people contacted an interpreter, either by attending Nature Tales, our formal interpretive programs, or by speaking with an interpreter on site. Because our site is a recreational draw for summer visitors, our programs need to be both educational and entertaining; while entertaining visitors, we slip them some knowledge.

Where does conservation fit with all this? Conservation is generally not a "fun" topic. It's hard to be light and bubbly while talking about a species on the brink of extinction. If you are too dour and serious, few people will take time to stop and watch your program. And yet, how can you talk about Siberian tigers, with less than 600 in the world, in a light way? And the more you research, the more saddened you are by the situation.

Traditionally, the zoo has tried to hit people with the conservation message (a required element) at the end of the 20-minute program, so that they've had 19 minutes of fun and interesting facts, maybe sung or acted a bit, and then 1 minute of seriousness. However, the audience is now expecting the "hit" at the end and they tune out or walk off.

Our new approach is to try to weave a conservation message into the whole program. For instance:

- Wanda Whooper is desperate to find a mate, because there are so few male whoopers for her to chose from.
- Tigger finds out he's not the only one, but there's not too many of his kind of critter.
- Why are there harbour seals in Calgary? (They're rescue animals.)
- Exploring some of the theories about dinosaur extinction.
- Why it's so important that our western lowland gorillas are breeding, and that 1 female in particular has had more youngsters than the other 2 females.
- How our Asian elephants came to Calgary—2 were orphaned in the wild.
- Rocky Mountains 2020—a futuristic approach: what the Rockies will be like in 2020, and why grizzlies will only be seen at the zoo in the future.

CONCLUSION

Education is a part of conservation. Most people honestly don't know much about animals and the ecosystems they are found in. Our aim is to help excite some interest, and we are always happy to talk for hours to interested visitors. Television nature shows are often too heavy or too light for most viewers, and you can't ask questions afterwards. We try to provoke visitors, so they have to ask questions like:

- Aren't Siberian tigers white?
- Are the enclosures big enough?
- Are they bored?
- What are the elephant trainers doing?

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- What do you feed them? Where does it come from?
- Do you go in with the animals? Are they dangerous?
- Why aren't there more babies at the zoo?
- What happened to the petting zoo?
- Where's the rhinoceros?
- Where do you get your animals?

We welcome these kinds of questions because it means our visitors are thinking about the zoo. Questions like these give us an opportunity to share more information with visitors than we normally do. As a new zoo, we should be confident enough to invite questions, so that we, and our visitors, can grow.

Sustainable Restoration? A Community Perspective on Environmental Capacity-Building

B. Bauer[1]
The Steelhead Society and Habitat Restoration Corporation

ABSTRACT

As recognition of the importance of healthy, functioning ecosystems grows, so does the perceived need to mitigate environmental destruction. To this point many restoration interventions have been largely orchestrated and carried out by specialist organizations non-resident in the local community. Using the experience of the Steelhead Society and Habitat Restoration Corporation (SSHRC) and its partnerships with First Nations and community organizations across British Columbia as examples, this paper looks at the significance of local involvement, "a community legacy", in restoration projects as an integral element of process planning. Specifically, the significance of Delgamuukw and the current British Columbia treaty process, and the resultant growth in First Nations' expectations for inclusion in partnerships is examined. This paper argues that First Nations are ideally positioned to take on leadership roles in the development and implementation of restoration and conservation projects. As well, the SSHRC's watershed restoration project management experience has shown that any environmental group operating within First Nations' traditional territories will increasingly have to contend with the expectation that project implementations will contain a meaningful element of capacity-building for First Nations communities.

[1] Present address: 103 - 131 Water Street, Vancouver, BC, V6B 4M3
info@steelheadsociety.com

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

A Case History of Community-Based Involvement in the Management of a Species at Risk: Wintering Bald Eagles in the Squamish Valley

Barry P. Booth
PAW Research Services
2822 W. King Edward Avenue, Vancouver, BC, V6L 1T9, Canada
bbooth@portal.ca

Markus Merkens
PAW Research Services
8463 12th Avenue, Vancouver, BC, V3N 2L8, Canada

ABSTRACT

This paper discusses the results of a research project that examined the wintering bald eagle (*Haliaeetus leucocephalus*) population in the Squamish Valley, British Columbia. It also examines the way in which the community incorporated and modified the recommendations of the research project to suit its needs. The Squamish Valley has long been known as a major wintering ground for bald eagles in North America. The many eagles that come to the valley each winter have attracted attention from all across North America and the world. In recent years there was a groundswell of concern regarding the health of the wintering population as records from midwinter counts suggested a downward trend in eagle numbers. Spurred by this concern, the Nature Conservancy of Canada initiated a research project, later funded by Forest Renewal BC, that explored the distribution and seasonal changes in eagle numbers in the valley over 3 winters. The results demonstrated seasonal shifts in eagle numbers, identified numerous areas of eagle concentration in the valley, identified potential threats to the winter population, and suggested actions to curtail these threats. The citizens of Squamish and Brackendale, with the help of the Nature Conservancy of Canada, adopted many of the recommendations from the research project. These community-based initiatives have attempted to incorporate bald eagle wintering habitat needs with present and future commercial and noncommercial activities.

Key words: bald eagles, community, *Haliaeetus leucocephalus*, Squamish, volunteer.

During October and November each year the freezing of interior waterways (lakes and rivers) drives many bald eagles from their summer habitat to areas with open water and a readily available source of food. Areas in southwestern British Columbia that provide these conditions have long been recognized as major bald eagle wintering habitat (Stalmaster 1987). Of the wintering areas in this region, the Squamish River valley has been known to support the largest documented concentration of eagles (Farr and Dunbar 1988). In the case of the Squamish Valley, this phenomenon is occurring in a region that has, and is undergoing, significant human-induced changes.

The many eagles that come to the valley each winter have attracted attention from all across the world. In recent years, as midwinter counts have suggested a downward trend in eagle numbers, a groundswell of concern arose regarding the health of the wintering population. This concern and others spurred the Nature Conservancy of Canada (NCC) to initiate a research project, later funded by Forest Renewal BC (FRBC), that explored the distribution and seasonal changes in eagle numbers in the valley. A secondary, and unanticipated, component of this project examined the potential impact and mitigation of eco-tourism on the winter eagle population.

STUDY AREA

The study was conducted approximately 70 km northwest of Vancouver, B.C. (Fig. 1) and includes three main rivers: Squamish, Cheakamus, and Mamquam. Lowland valley bottoms near the Municipality of Squamish are a combination of private, provincial Crown land, and First Nations reserve lands. Private land and First Nation lands also extend up the Squamish and Cheakamus River valleys. A large portion of the upper Squamish River lies within Tree Farm Licence 36. Forests in both lowland valley bottoms and upland slopes throughout the study area have been harvested considerably. In addition, much of the lowland valley bottom habitat near the communities of Squamish and Brackendale has been converted to residential and commercial uses. An extensive

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dike system to control flooding is also maintained along all three major rivers within the Municipality of Squamish. Coho (*Oncorhynchus kisutch*), chum (*O. keta*), pink (*O. gorbuscha*), chinook (*O. tshawytscha*), and sockeye salmon (*O. nerka*) spawn within the bounds of the study area. Several salmon-bearing streams in this system have, and are undergoing, extensive rehabilitation through the Watershed Restoration Program of FRBC. Two hatcheries produce both chum and coho salmon on the Cheakamus River.

METHODS

Daytime Eagle Censuses

Bald eagles were surveyed along 76 km of river in the study area: 61 km along the Squamish River, 11 km along the Cheakamus River, and 4 km along the Mamquam River.

Figure 1. Location of Squamish Valley wintering bald eagle study.

Surveys were conducted on foot and from helicopter, canoe, and commercially guided river rafts. During each census the number and location of eagles relative to the main channel of the river were recorded on data sheets consisting of maps traced from recent aerial photographs.

To examine the change in eagle distribution within years, we divided the rivers surveyed into kilometre sections. We then summed the number of eagles that occurred within each kilometre section in each survey week. Eagle numbers occurring in each kilometre section were plotted within 7 eagle density classes (number of eagles/km): 1–10, 11–25, 26–50, 51–100, 101–200, 201–400, >400 for the entire study area.

RESULTS

Population Dynamics

Bald eagles wintered in the Squamish Valley from mid-November to mid-February. Eagle counts ranged from 363 to 2,575 in 1996–97 and from 518 to 3,976 in 1996–97. Eagle numbers were typically low early in the survey season, peaked in mid-December, and then declined towards the end of the survey season (Fig. 2). Our peak counts of eagles in both 1995–96 and 1996–97 far exceed wintering eagle populations in the lower 48 U.S. states (Stalmaster et al. 1979, Lingle and Krapu 1986, Issacs and Anthony 1987, Isaacs et al. 1996, Stalmaster and Kaiser 1997) and all other known areas in southwestern British Columbia (Farr and Dunbar 1987). These numbers indicate that the Squamish Valley represents an extremely important bald eagle wintering area in the southwestern region of British Columbia and the U.S. Pacific Northwest.

Eagle Distribution Patterns

Eagle distribution throughout the valley showed consistent patterns between years (Fig. 2). Early aggregations of eagles occurred at the confluence of the Squamish and Cheakamus rivers, and the Mamquam and Squamish rivers in both winters. As winter progressed, eagle numbers increased near the confluence of the Squamish and Cheakamus rivers in both years. Congregations of eagles were also noted at the confluence of the Squamish River and Ashlu Creek and areas along the upper-Squamish particularly at the mouth of Shovelnose Creek. During the peak survey period (Survey #3 in both years) eagles reached extremely high densities along a 10-km stretch near the confluence of the Squamish and Cheakamus rivers. Other areas of eagle concentration were near the Ashlu Creek/Squamish River confluence, as well as the confluence of the Mamquam and Squamish rivers. A relatively long section of the Squamish River (between the confluence with Ashlu Creek and the Cheakamus River) rarely supported significant numbers of eagles.

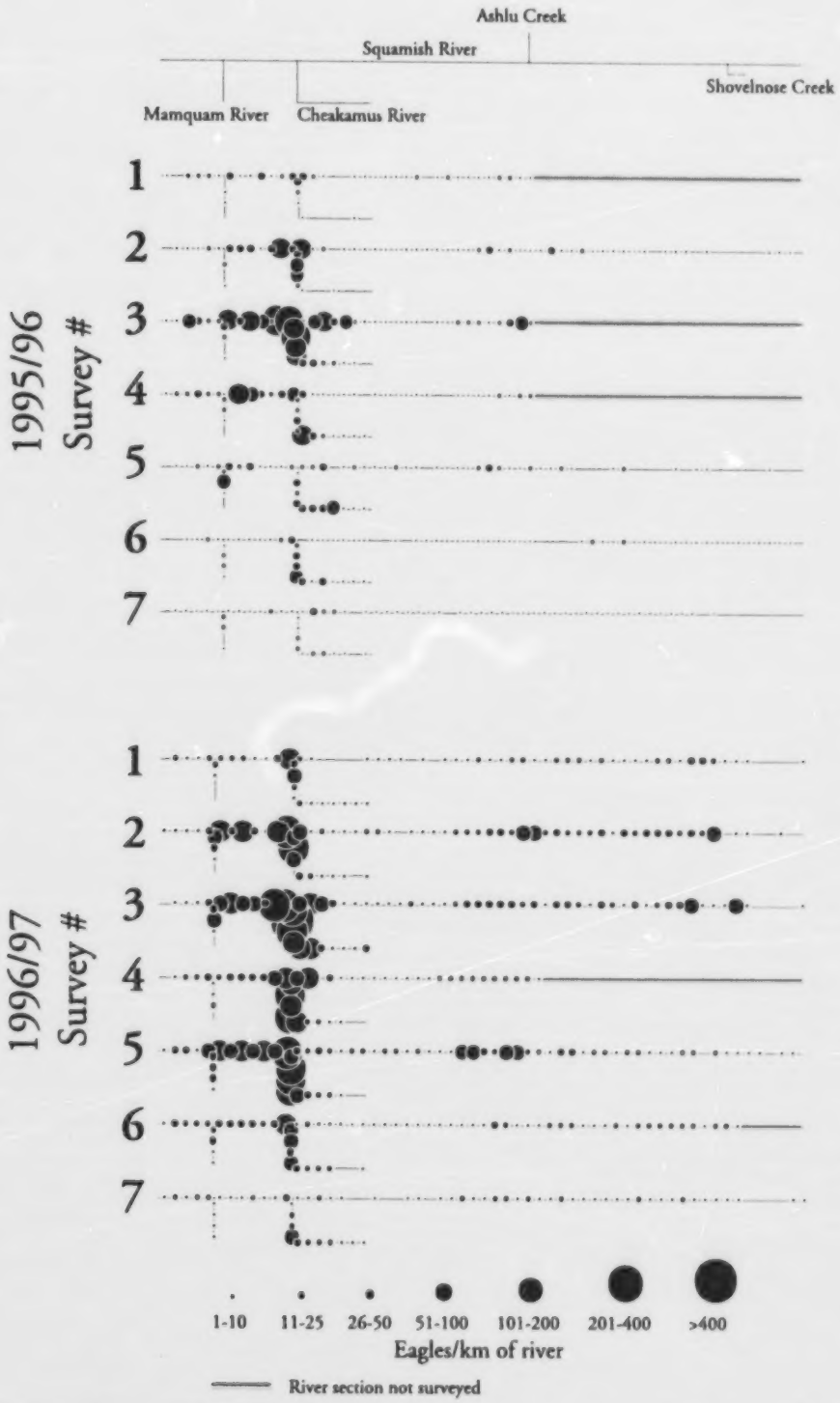

Figure 2. Number of bald eagles counted during each survey week in the Squamish Valley during the winters of 1995/96 and 1996/97.

Effects of Disturbance from the Eagle-Viewing Public

During our field research we encountered numerous disturbance events in which a significant number of feeding and perching eagles flushed in response to human activity. We also noted distinct differences in the distribution of eagles in response to human disturbance. During periods of high human disturbance (usually some weekends), eagles often avoided areas that supported numerous eagles during times of low human disturbance (generally weekdays). We documented disturbance from commercial river rafting, kayaking, and walking tours; individuals or groups watching the eagles by car and by foot; anglers; and local residents jogging or walking both with and without dogs. Of concern was that these events often occurred in areas of high eagle activity, particularly feeding areas (gravel bars). Human access to gravel bars and riverside deciduous perching habitat is possible by the network of dikes along the Squamish and Mamquam rivers, and along a stretch of railway track along the Cheakamus River.

DISCUSSION

Eagle Population

Results from our study clearly showed that extremely large congregations of bald eagles were centred around a relatively small stretch of river. This is due largely to the extensive network of gravel bars along a 10-km stretch of river near the confluence of the Squamish and Cheakamus rivers. This network of gravel bars represents focal points for the deposition of salmon carcasses from wild runs and the numerous enhancement activities along the Cheakamus River. A similar situation exists near the mouth of the Mamquam River where it meets the Squamish River. The gravel bars at the mouth of the Mamquam River, and those farther downstream along the Squamish River, receive salmon carcasses that are washed down from several small side channels and salmon enhancement areas along the Mamquam River. Areas of eagle congregation in the upper Squamish River are associated with side channels and areas with extensive salmonid enhancement activities.

Human Disturbance

Our study suggested that a visitor management program should be instituted to mitigate the disturbance that the eagle-viewing public was having on the wintering bald eagle population. We believed that it was necessary to initiate such a program due to the evidence collected in other wintering areas showing that wildlife viewing, as well as other human activities, can negatively affect wintering bald eagles (Stalmaster and Newman 1978, Skagen 1980, Stalmaster et al. 1991). In some areas of the western United States where human disturbance has been deemed excessive, the timing and extent of human traffic in eagle wintering areas have been restricted.

From our observations, and research elsewhere, we developed a management plan to reduce or mitigate the amount of human disturbance in the area. We identified 4 points of action:

- identifying foot and vehicle access points that need eagle-viewing information signs;
- identifying areas that could benefit from eagle-viewing platforms or kiosks;
- developing one or more types of eagle interpretative signs (i.e., "do not enter" signs and "view with care" signs); and
- considering a volunteer bald eagle warden program whereby local residents patrol high eagle use areas and provide information on eagle biology and viewing ethics.

Response to Research Recommendations

Eagle Watch Volunteer Interpretative Program (EW)

In response to our recommendations, the NCC approached some members of the communities of Squamish and Brackendale, particularly the Squamish Estuary Conservation Society (SECS), to determine if sufficient interest existed within the community to initiate a volunteer interpreter program. In the winter of 1995–96, the Eagle Watch Volunteer Interpreter Program (EW) was established to educate the viewer about eagle biology and viewing ethics; inform visitors about community services; monitor the number of eagle viewers; and provide information about the proposed Eagle Reserve (Porter 1996). Starting in 1996, EW positioned volunteer interpreters from 0900 to 1500 hours during the weekends of January and February near one of the best and least disruptive sites for viewing eagles in the Squamish Valley. Since then, EW has started in mid-December and has run to mid-February.

As EW has grown, so have the materials that it has used as instructional aids. In the first year, EW interpreters used portable educational signs in their interpretative duties. Since then a permanent structure has been built that houses educational materials regarding eagle biology as well as shelters for interpreters and the eagle-viewing public. Additional signs have also been constructed that outline the relationship of the Squamish Nation to bald eagles and other aspects of the natural history of the region. All of the interpretative signs and the shelter were designed in the community.

Make-Up of EW Participants and EW Accomplishments

The EW program is made up of volunteers from the local community. EW also includes 1 or 2 paid coordinator(s) annually. Initially most of the volunteers were members of the SECS. Since the first year, volunteers have come from both the SECS and the general public. Volunteer participation has increased significantly since the inception of the program (29 volunteers in year 1, 40 volunteers in year 3). The rate of return has also been a high—in fact almost 100% of the volunteers in year 3 returned to volunteer in year 4. This increase in public participation, largely due to an aggressive

recruitment drive by EW coordinators, has included numerous incentives from locally based businesses (e.g., discounts at local businesses, complimentary rafting trips). In addition, EW coordinators have used innovative approaches to attract volunteers. During one winter season volunteers came to the EW via a youth program that allowed students to work in the community (EW) in exchange for financial credits toward post-secondary education.

For the 3 years that data are available, EW interpreters have directly contacted over 8,500 eagle viewers. The Squamish Nation has also acknowledged the role that EW has played in directing excessive eagle-viewing traffic away from one of their more populated reserves.

Project Support: Political and Financial

Nature Conservancy of Canada

The NCC has played a pivotal role in the formation and persistence of the EW program. It has provided direct financial support and augmented financial input to the EW program via fundraising both within and outside the community. Major accomplishments by the NCC include constructing the eagle-viewing shelter and its associated interpretative materials, negotiating the transfer of the shelter to the District of Squamish (DoS), and making direct financial contributions to the day-to-day operations of EW. The NCC has also commissioned additional eagle interpretative signs for specific locations within the Squamish Valley. While the NCC has contributed immensely to the EW program, it has always believed that the community should retain ownership and control of the project.

Squamish Nation

Individuals from the Squamish Nation have not directly participated in the EW interpretative program. The Squamish Nation has, however, participated in EW in different and important ways. Carvers from the Squamish Nation joined with students from Howe Sound Secondary School to produce carvings later used in the EW shelter. In addition, members of the Squamish Nation have contributed interpretative information for signs that depict First Nations' relationship with eagles. Individuals from the Squamish Nation, including elders, have also addressed training sessions for interpreters to pass on the knowledge and wisdom of the relationship between the eagles and the Squamish Nation to coordinators and volunteers. Blessing ceremonies have also been conducted for the EW shelter and its carvings. Recently, the Squamish Nation has been involved in a literacy project that incorporates components of the EW program. An educational book and CD-ROM are a planned part of this endeavour.

Provincial Government

Neither the Ministry of Environment, Lands and Parks, nor the Ministry of Forests has directly participated in the EW program. In contrast, the U.S. Forest Service in Washington State provides a direct financial contribution, as well as support staff in a cooperative program with the North Cascade Institute in a similar interpretative program on the Skagit River.

Municipal Government

The DoS has provided logistical and in-kind support to EW. It has facilitated the provision of parking near a permanent visitor shelter and issued permits for construction of the EW shelter. The DoS has also accepted responsibility for the maintenance of the shelter, constructed an on-site toilet, and provided secretarial support for EW report writing. The DoS has not made direct financial contributions to EW. The lack of direct financial support is due to several reasons: the community is experiencing financial difficulties; district council does not believe that EW attracts significant business to the local area during the winter season; and council has not been satisfied that EW has the support of the entire community. The result is that the DoS does not believe that EW merits financial support from the local government. The local chamber of commerce (COC) has voiced a similar sentiment. As a result, the DoS and the COC have indicated that future financial contributions or assistance in long-term planning will not be forthcoming.

Local Businesses

Little direct financial support for EW has come from the small business community. Rather, some of the local businesses in the Squamish area have contributed to the EW program by providing in-kind support (e.g., river raft trips and discounts at local shops for EW volunteers, production of promotional material). The lack of direct support may be due to an economic downturn in the area; competition for a limited pool of charitable dollars (e.g., local businesses have a large demand placed upon them by various community groups); and the perception that EW does not have broad community support.

Corporate Support

EW coordinators have secured financial contributions from several large corporations or businesses. With one exception, funding has been a one-time donation. At present no one business has made a long-term commitment to the EW program.

Long-Term Outlook for EW

The continued existence of the EW program in its present form is uncertain. Much of the financial support for EW has come from NCC. This financial support has always been viewed as temporary and has now ended. As a result, financial solvency of the program will become the full responsibility of the community. Efforts for more community assistance with program operations have been somewhat successful. Some businesses view the program as an independent organization. As a result, it has not garnered broad support from

the local business community. In addition, the program has not secured long-term financial commitment from any one, or group of, corporations. Furthermore, the responsibility for the continued existence of this program appears to rest in the hands of a few individuals within the community.

The long-term survival of EW will hinge on a variety of circumstances. It has been made relatively clear that EW will not gain additional support from the business community if it remains as a stand alone program that functions as an independent organization. To achieve the support of the community as a whole, EW must embrace the community more closely. One option is to incorporate the EW program with other programs that presently exist within the community. Plans to develop a winter festival that highlights more than just eagles have been discussed.

The EW program can increase local government and business support by demonstrating that tangible economic benefits are derived from the services that EW provides for the community in general. The EW program has taken steps this year to address this concern. EW volunteers have administered questionnaires designed to document the spending habits of the eagle-viewing public.

Long-term financial support may come from larger corporations. However, outside corporate support may be less desirable from a community perspective. Community ownership of the program could be diminished if financial support is derived from outside sources.

The EW program represents an ambitious project that has attempted to bring together a wide variety of community members for a common purpose. The project has struggled financially since its inception, but has achieved considerable success. Its enthusiastic volunteers have had contact with thousands of eagle watchers; it has demonstrated tangible results, especially in reducing eagle-viewing traffic within Squamish Nation reserve lands; it has incorporated First Nations participation; and it has partial local government and business support, and some financial support from larger corporations. EW, despite its problems, represents an important project that has implications for other similar programs.

CONCLUSION

Lessons from EW Applicable to Other Community-Based Projects

- Long-term funding is a priority and funding can be difficult. Partnerships with non-governmental organizations, such as the NCC, have been valuable in providing seed money and expertise that may not be present in every community.
- A program such as this needs widespread community support (local government, chamber of commerce, local businesses). From this support, funding from within the community has a greater chance of success.
- For a program to have widespread support within the community, it may need to be associated with other events in the community. One challenge will be to maintain an independent identity for the program should it become associated with other events.
- Support for such a program may also have to demonstrate a clear economic benefit to the community.
- Corporate funding may be welcome but concerns about community ownership being usurped should be considered.
- Alternative approaches to raising funds should be considered (e.g., approaching foundations, considering endowments).
- Volunteers are essential and must remain happy and willing to return. Ensure that volunteers are not overworked, and that their work is appreciated.

ACKNOWLEDGEMENTS

We thank the following people for taking the time to talk to us, thereby making this paper easier to write: M. Fellows (DoS), J. Porter (past EW coordinator), K. Davis (NCC), and D. Harry and E. Nelson Moody (both from the Squamish Nation).

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Return of the Peregrine Falcon

Mary Krupa
Morningstar Enterprises
212-1708 Dolphin Avenue, Kelowna, BC, V1Y 9S4, Canada
mkrupa@silk.net

ABSTRACT

Community action to restore the peregrine falcon to the Okanagan Valley was initiated exclusively by private funds. Nine young were released in 1998, and more are planned for subsequent years. A spin-off public environmental education campaign on endangered spaces and species directed especially at school children includes celebrities and a broad selection of sponsors and supporters.

Key words: *Falco peregrinus anatum*, Okanagan, peregrine falcon, reintroduction.

The Okanagan Valley, B.C. is 1 of the 4 most endangered ecosystems in all of Canada. It is also 1 of Canada's favourite tourist and retirement destinations. More and more of the valley's unique habitat is being developed every day. Among the many threatened and endangered species in the valley, few are as captivating as the *anatum* peregrine falcon (*Falco peregrinus anatum*).

A unique and amazing project is sweeping the Okanagan. Following the example of the Canadian National Peregrine Recovery Plan, 9 young peregrine falcons were released in July 1998 from the roof of Landmark II in Kelowna. The plan is to release birds for at least 2 more years in the hope that we can bring the magnificent *anatum* peregrine back to breed in the valley it once inhabited extensively. This is the first project of its kind in British Columbia, and the only peregrine recovery project in Canada to be initiated exclusively by private funds. The successful reintroduction of this species in British Columbia will complete *anatum* peregrine recovery in Canada from coast to coast.

Upon researching the natural history of the falcon in the Okanagan we learned that this valley had once been home to several pairs of *anatum* peregrines, but no breeding pair had been seen here since 1959. Organochlorine residue (DDT) was the primary factor in the deterioration of the *anatum* peregrine falcon population, not only locally, but throughout North America.

Falcons adjust very well to human encroachment and, based on this fact, their chances of survival are extremely high. In an effort to reintroduce these birds, a method known as hacking was developed. This procedure has been incredibly successful in almost every major city in Canada, but had never been attempted in British Columbia. Young falcons are placed in a "hack box" to help protect them from predators and the elements, and are introduced onto the top of a high building, man-made tower, or, very occasionally, a cliff. "Hack attendants" feed the birds through a chute. The birds are then released, instinctively learn to fly, and eventually migrate south. A small percentage of the birds return to the general location from which they fledged and establish a territory.

The Canadian experts in peregrine recovery advised us to conduct this project in an urban setting rather than at a natural cliff site because of the large owl population in the valley. The young falcons, without the protection of adult birds, make easy prey for owls and many other predators. Urban recovery projects have proven to be effective in other Canadian cities.

DESCRIPTION OF EDUCATIONAL ACTIVITIES

The peregrine falcon has become the focal point of a valley-wide public environmental education campaign. A series of 3 educational units on endangered species and spaces of the valley for grades K–3, 4–9, and 10–12 have been created with Vancouver Foundation funding. These are integrated with the British Columbia school curricular objectives and are available in French or English. These units will reach more than 30,000 students in 3 school districts this year, and act as a follow-up and reinforcement to live interpretive presentations done by a renowned local British Columbia Parks ranger, Blake Dixon. Blake and his falcon Nikita will visit more than 40 schools throughout the Okanagan over the next few months. This spring the project will feature a "Name the Falcon" contest, an Okanagan Endangered Wildlife art contest, and a Robert Bateman National Wildlife Week writing contest. These educational contests allow the students to become actively

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

involved in the project. Next fall we will also be providing schools with materials and professional assistance to apply for funds from agencies such as Habitat 2000 and the Evergreen Foundation to create and/or restore habitat on their school grounds.

To provide a focus for all of the positive action these educational activities are generating, we will be hosting an Endangered Wildlife Festival in Kelowna this spring. The honorary chairman, Robert Bateman, and the British Columbia Minister of Environment, Cathy McGregor, will both be attending. YTV has confirmed their support of the event and is acting as the exclusive national television sponsor. This will enable us to bring the educational opportunities of the festival to youth across Canada and to provide national recognition for festival sponsors. YTV began running 30-second commercials about the event in February. Sun FM 99.9, a very popular radio station in the valley, has committed to becoming the exclusive radio sponsor, and the Kelowna *Daily Courier* will be the print sponsor. An art festival, habitat creation/restoration contests, field trips, and nature exhibits will be among the many educational activities this festival has to offer. The winners of all educational contests will be personally congratulated by Robert Bateman at this event.

A documentary—*Return to the Valley*—has been created by Shaw Cable exclusively for this project. It has been entered by Shaw for the prestigious Omar Girard award. The video provides an excellent overview of what the project has accomplished in Kelowna. (Copy available upon request).

SUMMARY

The Return of the Peregrine Falcon is an innovative project that is having a discernible impact on our human and natural community. It is a catalyst for positive environmental change throughout the Okanagan Valley. Our team is lead by distinguished Canadians including Robert Bateman, the honorary chairman, and Gordon S. Court, leading Canadian peregrine specialist and scientific advisor to the project. David Suzuki's office recently called us requesting an essay from Kelowna students to be used in his upcoming book.

The Return of the Peregrine Falcon project is about far more than bringing back the peregrine—it is about learning what we can do to protect other wildlife species in the valley that are at risk. It promotes learning about what we can do to make the Okanagan a healthier place for all living creatures. Its key objectives are:

- To establish a breeding pair of peregrine falcons in the Okanagan by the year 2000.
- To educate the public, especially children, about the importance of wildlife conservation and the need to create a sustainable balance between the needs of human and wildlife communities. ("I cannot imagine a more perfect tool than the peregrine falcon to capture the imagination and hearts of young people." – Robert Bateman, Honorary Chairman).
- To help conserve rare and endangered wildlife and wildlife habitat in the Okanagan Valley for generations to come.

It is our goal to have several breeding pairs of falcons living in the valley they once inhabited extensively and to have the community actively involved and dedicated to wildlife conservation.

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999.

Laskeek Bay Conservation Society: 10 Years of Experience With Long-Term Monitoring, Education, and Advocacy as a Volunteer Organization

Charlotte L. Husband
Laskeek Bay Conservation Society
Box 867, Queen Charlotte City, BC, V0T 1S0, Canada
capeball@qcislands.net

Chris J. Bowman
Laskeek Bay Conservation Society
Box 867, Queen Charlotte City, BC, V0T 1S0, Canada

ABSTRACT

Laskeek Bay Conservation Society, which was established 10 years ago, is a small volunteer organization based on Haida Gwaii that runs a field program and diverse long-term monitoring, research, and interpretation programs.

Key words: community, Haida Gwaii, Laskeek Bay Conservation Society, nonprofit, volunteer.

A few like-minded friends around a kitchen table started the Laskeek Bay Conservation Society 10 years ago on Haida Gwaii/Queen Charlotte Islands. Our initial focus was to continue the work of Dr. A. J. Gaston in monitoring the ancient murrelet population in the Laskeek Bay ecosystem. A decade later we are involved in several important projects to protect endangered species and their habitat on Haida Gwaii which includes several Red- and Blue-listed species such as the ancient murrelet, marbled murrelet, Cassin's auklet, and Peale's peregrine falcon. As well we are active partners in major programs to address the serious impact of introduced species (e.g., Sitka black-tailed deer, racoons, and red squirrels) on the native and endemic species of the Haida Gwaii Archipelago.

This paper briefly describes the area where Laskeek Bay Conservation Society (LBCS) works, and some of our long-term ecosystem monitoring programs and then turns to how LBCS has linked these programs to public education and involvement in the conservation of rare ecosystems.

We are a small volunteer organization based on Haida Gwaii. From early April to mid-July of each year we run a field program from our research station on 50-ha East Limestone Island, in the Laskeek Bay area. We carry out diverse long-term monitoring, research, and interpretation programs in the surrounding islands and waters of Laskeek Bay.

When we began LBCS, our intention was to continue a research project started by Dr. A. J. Gaston of the Canadian Wildlife Service (CWS) on ancient murrelets *(Synthliboramphus antiquus)*. This beautiful little bird spends its life at sea, only coming to shore for a brief period each year to nest in the thousands beneath the roots of large trees on small offshore islands of Haida Gwaii, like Limestone and Reef in Laskeek Bay. About 60% of the world population of ancient murrelets nest on Haida Gwaii, the only nesting sites in Canada.

We felt that some of the interesting questions were only beginning to be asked. We wanted to turn this into a long-term study and talked of 25 years of monitoring to understand ancient murrelets and their interaction in the Laskeek Bay area. So, we convinced CWS and Canadian Parks Service to give us a couple of small contracts—LBCS was born.

We started doing long-term ecosystem monitoring to look at environmental changes and believed strongly that conservation-minded volunteers could do field work using low-technology methods. We felt that lack of accessible information was a barrier to incorporating raw data into working practice. We also were concerned that local communities had previously had little involvement in major research and inventory projects on Haida Gwaii. We did not find out the results of the projects and sometimes important management decisions that affected our communities were drawn from those projects.

So, LBCS set out with 4 purposes:

1. **To continue long-term monitoring of the marine and terrestrial ecosystems of the Haida Gwaii Archipelago.**

At the Limestone Island field station, staff researchers and trained volunteers continue to monitor the ancient murrelet population. This is one of the longest running population monitoring studies in Canada with 15 years of data. Dr. A. J. Gaston's book *The Ancient Murrelet* (Gaston 1992), and our annual scientific reports from 1990 to 1997 (Gaston 1991–98) describe fully the results of this work.

This data, combined with other research by the British Columbia Ministry of Environment, Lands and Parks (MELP) and CWS, led to the listing of the ancient murrelet as vulnerable nationally (Committee on the Status of

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Endangered Wildlife in Canada) and Blue-listed provincially (Hartman et al.1997). This listing is primarily due to the impacts of introduced raccoons and rats to the Haida Gwaii ecosystem and other environmental factors like oil spills.

Over the years, we have expanded monitoring activities to include other research projects. For example, our at-sea surveys of marbled murrelets are the longest running year-to-year survey/census of this endangered species. We feed statistics on marled murrelet into provincial and international databases. Other research projects we conduct are:

i) *Seabirds and shorebirds:* censusing black oystercatchers and gulls, monitoring Cassin's auklet nest sites in Laskeek Bay ecosystem.
ii) *Forest birds:* monitoring cavity nesters such as red-breasted sapsuckers and ravens. With other researchers we monitor songbirds before and after nesting to compare breeding success rates between islands on which squirrels have been introduced and those which are squirrel-free.
iii) *Other endangered species at risk:* peregrine falcon survey and nest site observation in Laskeek Bay ecosystem to monitor chick survival, fledgling success, and adult use of different nest sites.
iv) *Introduced species:* deer, raccoon, and squirrel and their impacts on native and endemic species and habitats. We collaborate on these projects with agencies such as Canadian Parks Service, CWS, MELP, and the Centre National de la Recherche Scientifique in France, and sit as a nongovernment organization (NGO) on the Working Group on Raccoon–Seabird Interactions (Martin and Daufresne 1996, Golumbia 2000, Harfenist et al. 2000).
v) *Marine mammals:* provide annual observation records and photos of orcas, Pacific white-sided dolphins, and humpback whales to researchers.
vi) *Rare and locally endemic plants and introductions:* An interesting phenomenon on Haida Gwaii is the general association between limestone rock and endemic and rare plants. Limestone rock is rare on Haida Gwaii, occurring only on a few islands, like Limestone Island, and at high elevations. A plant list begun in 1990 is updated annually. Deer are changing the plant community structure.

2. To provide opportunities for volunteers and students to participate in biological field programs and to offer training for necessary field research skills.

We assumed correctly that this training might lead to future employment or educational opportunities for local people. Volunteers gain experience and training by working with field staff monitoring the wildlife populations and learning data collection techniques. Some former volunteers now have jobs and careers in scientific research. In addition, we now have a student intern program for young people interested in a career in science.

3. To promote better understanding of marine and terrestrial ecosystems and the local and global influences that threaten the native species.

The purpose was simply to provide information about the biology of the Laskeek Bay ecosystem to locals and visitors, both at the field camp and in communities, through public outreach programs. A top priority is involving school children at the field camp and in their classrooms. While on Limestone Island, the volunteers and school children develop a deeper understanding of local island ecosystems, species, habitats at risk, and principles of environmental stewardship.

Our experience is that volunteers at Limestone Island become ambassadors in their own communities, sharing what they learned as participants, helping to change attitudes, and advocating conservation issues.

4. To support and assist other programs aimed at improving knowledge through research and monitoring; and at promoting better decisions in management and conservation of ecosystems by being a local partner and source of advice for various federal and provincial agencies.

We wanted to support and assist other programs on Haida Gwaii aimed at better management and conservation of species and habitats. In this regard, we wanted to be a local partner and a source of local advice for various federal and provincial government agencies and universities.

Ten years later, we have had successes in all 4 of our purposes.

1. We raised awareness by building a local support base via our volunteer and school programs.
2. We published information so that those people without a scientific background could understand. Tabloid newspapers were distributed throughout the local community.
3. In our case, an important issue turned out to be introduced species. Major environmental impacts are caused in our coastal zone by raccoons and rats that prey on seabird populations and from deer that eat vegetation, fundamentally changing the forest composition and habitats. We joined others in forming a coordinated research project on introduced species on island ecosystems called Research Group on Introduced Species (RGIS) which is now actively addressing these issues.
4. The government embarked on raccoon and deer control because a group of volunteers built a grass roots awareness promoting the need for management of the introduced species to protect the endemic species at risk.

LBCS used many strategies to accomplish our work over the last 10 years. We have the following strategies to offer to other NGOs for implementing in their own communities. This can effectively establish NGOs as active players in protecting rare and endangered species and habitats.

STRATEGIES FOR AN EFFECTIVE NGO

Involve volunteers

- As research assistants
- As ambassadors
- As board members
- As scientific advisors
- As interns

Educate

- Train staff through employment
- Invest in youth (go into schools, have children come to projects)
- Advocate through information
- Base all education products on facts learned in research projects
- Develop methods manuals
- Reach eco-tourists

Build partnerships

- Research scientists
- Government agencies
- Universities
- Museums
- A memorandum of understanding will clarify roles

Get funding

- Foundations
- Corporate sponsors
- Local business
- Private donations
- In-kind donations
- Contract management fees
- Youth and intern scholarship programs

Don't go beyond your resources

- Financial
- Area of operation, expertise, knowledge
- Volunteer abilities
- Nonconfrontational advocacy (don't burn bridges)
- Keep small and flexible (vs. large agencies)

Build credibility

- Be financially accountable
- Register as a nonprofit society
- Train staff
- Plan
- Develop an office infrastructure
- Publish articles, papers
- Join committees
- Network

Get your message out to a broad audience

- Newsletters
- Web sites
- Volunteers as ambassadors
- TV and news media
- Slide shows
- Public presentations with visiting experts or researchers

CONCLUSION

The LBCS works with the ancient murrelet and other species at risk. While carrying out our initial goal we found that the preservation of the ancient murrelet was contingent on several converging factors in the habitat including other species coexisting in the same ecosystem. We decided to improve and increase local knowledge about island ecosystems and species at risk to increase community participation in local long-term management decisions. LBCS has found that an NGO can, through some of the previous strategies, especially by using volunteers, creating jobs, and supplying educational programs, be a link between government, academic research, and the public. When the public is involved in such an organization or receiving the educational programs, individuals will be better informed to take ownership of the issues and thus support sound management decisions.

ACKNOWLEDGEMENTS

The Laskeek Bay Conservation Society is a nonprofit volunteer-run organization and could not operate without the generous support from a wide variety of groups and individuals. We gratefully acknowledge the contributions of all our supporters and apologize for any omissions to this list. The Science Advisory Committee, which consists of A. J. Gaston (CWS, Ottawa); J. L. Martin (CNRS, France); R. Butler (CWS, Delta); K. Heise (University of British Columbia); and T. Reimchen (University of Victoria) oversees LCBS research in collaboration with the Research Group on Introduced Species (RGIS), which also includes Archipelago Management Board (Parks Canada, AMB); MELP; Centre National de la Recherche Scientifique, Montpellier, France (CNRS); and National Wildlife Research Centre (CWS).

LBCS receives generous contributions from W. Alton Jones Foundation, the CWS, Haida Gwaii Museum, Gwaii Trust, South Moresby Forest Replacement Account, and most importantly from the society staff and volunteers.

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The Howe Sound Watershed Stewardship Project

D.G. Blair-Whitehead
Bowen Island Conservancy
Box D82, Bowen Island, BC, V0N 1G0, Canada

A. Finlayson
Bowen Island Conservancy
Box D82, Bowen Island, BC, V0N 1G0, Canada

W. Husby
Bowen Island Conservancy
Box D82, Bowen Island, BC,V0N 1G0, Canada
w_husby@istar.ca

ABSTRACT

In 1995 the project team was becoming concerned. The population of Bowen Island and the surrounding Howe Sound Basin was growing rapidly. Many newcomers were not aware of the complexity and importance of local watersheds or the impacts their actions have on their watersheds. We were concerned that the present population growth may not be sustainable. We decided that a broad local community education thrust was needed to help citizens understand their place in their watersheds and make informed decisions about development. The project's goal is to increase public awareness, understanding and positive action using watershed stewardship activities in the Howe Sound Basin. Target participants were people interested in watershed issues: residents and landowners, community groups, leaders and decision-makers, and staff of local government. Two phases are complete: the Pilot (February - June 1995) and Phase One (September 1995- December 1996). To date, the project has: developed, printed and distributed the community watershed stewardship manual, *Water for Tomorrow: A Guide to Watershed Stewardship in the Howe Sound Basin*; developed and delivered activity-based workshops in communities throughout the Howe Sound Basin including Bowen Island (twice), Gambier Island, Gibsons Landing, Squamish, and Whistler; encouraged the initiation of local stewardship projects by increasing participant knowledge of watershed ecology, management techniques, use of forest and water resources, communication and networking skills amongst key players in the watershed. Workshops and the manual have covered the following topics: Introduction to Watershed Stewardship; How a Watershed Works; Mapping Your Watershed; Tools for Stewardship; Making Watershed Stewardship Happen.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Kingfisher Creek Watershed Restoration: A Case History

J.P. den Dulk

EcoTec Environmental Consultants Inc.
2621 Golf Course Drive, Blind Bay, BC, V0E 1H1, Canada
ecotec-west@bc.sympatico.ca

ABSTRACT

The Kingfisher Creek watershed was chosen as a demonstration watershed by the British Columbia Ministry of Environment, Lands and Parks (MELP) and Forest Renewal British Columbia (FRBC) in order to advance the techniques utilized within the Watershed Restoration Program (WRP). It is located in the Interior Cedar–Hemlock biogeoclimatic zone east of Enderby, B.C., approximately midway between Calgary, Alberta and Vancouver, B.C. The impacts throughout this area include the removal of riparian vegetation, input of sediment, exaggeration of the hydrologic regime, loss of in-stream large woody debris (LWD), and disconnection of the main channel from its floodplain. Fish species affected include anadromous coho, chinook, and sockeye salmon; resident rainbow and bull trout; and adfluvial rainbow trout populations from both Mara and Mabel lakes. The restoration works completed include: the development of a side channel near the confluence with the Shuswap River to restore spawning and rearing habitat; up-grading of trails and fish access to the Cougar groundwater channel (Hunter's Creek) at 10.5 km, Forest Service Road (FSR); bioengineering of eroding banks and addition of LWD-boulder structures to Danforth Creek under the B.C. Hydro utility corridor at 12 km, FSR; addition of in-stream LWD and riffle structures to Danforth Creek adjacent to the access road and bridge at 16 km, FSR; and addition of LWD to Danforth Creek throughout the B.C. Hydro utility corridor at 18.5 km, FSR. Altogether, approximately 12,250 m^2 of over-wintering pond habitat and 2,670 m of rearing and spawning channel habitat were treated with restoration prescriptions. This project created 356 person-days of employment, and final expenditures amounted to $155,774.33.

Key words: fish, habitat restoration, large woody debris, *Oncorhynchus* spp., riparian, salmon, *Salvelinus confluentus*, trout, watersheds.

PROJECT BACKGROUND

In 1997, Forest Renewal British Columbia (FRBC) endeavoured to establish demonstration watersheds throughout British Columbia in order to provide opportunities for the training of restoration specialists in the various techniques already established by the Watershed Restoration Program (WRP). Through a roundtable established for this task, Kingfisher Creek was chosen as a representative watershed for the southern Interior region.

Restoration prescriptions were investigated, formulated, and implemented for 1 groundwater development project within the Kingfisher Creek watershed during the 1997 work season to address (primarily) lost rearing habitat features for coho salmon. This channel is located on Hunter's Creek at 10.5 km on the Three Valley–Mabel Forest Service Road (FSR). A side-channel diversion project was also initiated "in the dry" with the anticipation of completion during the 1998 fisheries-work window. At the completion of the 1997–98 fiscal year it was anticipated that future works would continue to focus on the habitat requirements of coho salmon while addressing other physical and biological issues within the watershed.

The process to determine specific prescriptions included the involvement of various staff from the British Columbia Ministry of Environment, Lands and Parks (MELP), FRBC, Department of Fisheries and Oceans (DFO), United States Forest Service, British Columbia Ministry of Forests (MOF), and B.C. Hydro, as well as members of the Spallumcheen Indian Band (SIB) and the Kingfisher Environmental Interpretative Centre Society (KEICS), several consulting firms, numerous sub-contractors, and 2 local landowners.

As a result of the work conducted in 1997, EcoTec Environmental Consultants Inc. was retained to design and implement the prescriptions for 1998. Jan den Dulk was assigned as project manager based upon his extensive experience conducting numerous fish habitat and channel condition restoration projects throughout British Columbia since 1996.

Works Completed in 1997–98

The project located at 10.5 km on the FSR is supplied by a groundwater infiltration gallery and utilizes an abandoned

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overflow channel on the Hunter's Creek floodplain approximately 0.5 km upstream from the confluence with Kingfisher Creek. The habitat produced includes approximately 650 m of flowing (channel) and 6,000 m^2 of still (pond) water. Large woody debris (LWD), riparian plantings, and the addition of spawning gravels all enhanced the habitat quality.

The project located near the confluence with the Shuswap River is supplied by a combination of natural hillside springs and an intake diversion from Kingfisher Creek, and utilizes an abandoned side-channel gazetted as Gonzales Creek. The habitat produced includes approximately 1.2 km of flowing and 6,000 m^2 of still-water habitat. Riparian plantings, the addition of LWD, and the establishment of gravel beds have augmented the over-wintering habitat and spawning features in this project.

WATERSHED CHARACTERISTICS

Kingfisher Creek is located 30 km east of Enderby, B.C. and encompasses 75,000 ha, which drain south into the Shuswap River downstream of Mabel Lake. The watershed includes both Danforth and Hunter's creeks, in addition to the mainstem river, and lies within the Interior Cedar–Hemlock biogeoclimatic zone. It has been modified through agriculture and livestock free-ranging, utility-corridor construction and maintenance, forest fires, and timber harvesting activities. This area is also heavily used for recreation and tourism.

Fish species that utilize the watershed include: anadromous coho salmon (*Oncorhynchus kisutch*), chinook salmon (*O. tshawytscha*), and sockeye salmon (*O. nerka*); resident rainbow trout (*O. mykis*) and bull trout (*Salvelinus confluentus*); and Mabel Lake and Mara Lake adfluvial rainbow trout populations. (Note that other species may be identified in the Overview Fish Habitat Rehabilitation Procedure [FIIRP] yet to be published.) The current run of mid-Shuswap coho is believed to be only a fraction of historical escapement numbers and has been designated as threatened (P. Slaney, B.C. Ministry of Environment, Lands and Parks, 1999, pers. comm.).

Adverse impacts on fish habitat throughout the watershed include the removal of riparian vegetation (floodplain timber harvesting); input of sediment to the creek (lateral channel avulsions and livestock trampling); exaggeration of the hydrologic regime (up-slope timber harvesting); loss of in-stream LWD (active removal and disintegration); and disconnection of the main channel from its floodplain (road/bridge construction and channel degradation). These are largely the result of past timber harvesting practices and utility corridor construction and maintenance.

PROJECT DESIGN

The restoration works proposed for 1998 focused on side-channel development and "in-stream" techniques aimed at the placement of LWD, bank bioengineering, and spawning and rearing habitat creation.

Works Proposed for the 1998 Fisheries-Work Window

Specifically, the 1998 prescriptions and work plan included:

1. Continued development of the side channel to restore spawning and rearing habitat near the confluence of Kingfisher Creek and the Shuswap River.
2. Upgrading the access trail system through the groundwater channel site at 10.5 km on the FSR and adding additional LWD to the rearing ponds.
3. Stabilization of the mainstem channel through a combination of hard and soft engineering techniques, from the confluence of Danforth and Kingfisher creeks (approximately 12 km on the FSR under the B.C. Hydro right-of-way) and extending upstream approximately 120 m on Danforth Creek.
4. Placement of in-stream LWD and boulder structures to provide channel stability and rearing habitat for coho salmon where the spur road at 16 km on the FSR crosses Danforth Creek.
5. Development of off-channel alcoves and placement of in-stream LWD to provide channel/floodplain stability and rearing habitat for coho salmon under the B.C. Hydro right-of-way at 18.5 km on the FSR.

IMPLEMENTATION SUMMARY

Due to extremely low flow conditions in 1998 the majority of in-stream works were, in effect, completed in-the-dry, posing no risk of sediment discharge or disturbance to resident fish populations. As a precaution, cross-stream silt-retention fences were established downstream of all sites for the duration of the construction. All works were initiated on or after 10 August 1998 and completed on or before 10 September 1998. The works completed after the close of the fisheries-work window (31 August) were limited to riparian planting, trail upgrading, and other activities outside of the wetted channel. All restrictions, conditions, and exclusions issued by either MELP or DFO were strictly adhered to and no deviations from the approved work plan were requested.

Table 1. Erosion-control grass seed mixture applied to disturbed sites.

Common name	Species name	Percentage
Perennial ryegrass	*Lolium perenne*	15
Alsike clover	*Trifolium hybridum*	10
White Dutch clover	*Trifolium repense*	10
Red clover	*Trifolium pratense*	5
Timothy	*Phleum pratense*	15
Meadow foxtail	*Alopercurus pratensis*	10
Kentucky bluegrass	*Poa pratensis*	5
Canadian bluegrass	*Poa compressa*	5
Chewings fescue	*Festuca rubra* var. *commutata*	10
Creeping red fescue	*Festuca rubra*	10
Wildflower lupines	*Fabaceae* sp.	5

CHRONOLOGICAL SUMMARY OF ACTIVITIES

Planning meetings to review the proposed prescriptions and work plan for all sites were initiated in June and July. Those in attendance included representatives from MELP, MOF, DFO, Riverside Lumby Ltd, SIB, KEICS, B.C. Hydro, and FRBC. Key technical input was received from P. Slaney, W. Koning, and R. Finnigan (MELP, Fisheries Branch), B. Symonds and D. Gooding (MELP, Water Management Section), and R. Tomich (FRBC).

Davison-Tomich Side Channel

Development of the Davison-Tomich side-channel (gazetted as Gonzales Creek) was initiated on 10 August 1998. This channel is on private property owned by K. Davison and R. Gerry, and letters of agreement for these works are on file with MELP in Kamloops. Prior to construction, this site constituted early second growth (<25 years old) on a poorly sorted, imperfectly drained alluvial fan. Vegetation was primarily birch-alder with evidence of an emerging young cedar-fir understory. Gonzales Creek was an isolated side channel and had been in-filled by periodic flood events (the latest of which occurred on 6 July 1997), to produce a fen with no surface flow or connection to the mainstem channel. Numerous natural springs supplied groundwater from the hillside to the west, and infiltration from Kingfisher Creek supplied seepage from the east.

Using an Hitachi EX 270 excavator, an intake was established to Kingfisher Creek approximately 650 m upstream from the confluence with the Shuswap River. This location presented a relatively stable reach of the mainstem channel on the outside of a naturally maintained meander. The intake flows into a settling pond, which is protected from bankfull-flood-return periods by an existing vegetated berm. An adjustable head gate controls the flow leaving the pond through 220 m of 0.35-m-diameter, ribbed PVC pipe running to the excavated channel. The completed project constitutes 1.2 km of channel (average 1 m wide, 0.5 m deep, and 0.75% slope) lined with 10–15 cm of river rock, and 6,000 m^2 of pond area (average 1 m deep) with approximately 40% LWD surface coverage.

Table 2. Native shrubs planted adjacent to the Mabel Lake Road.

Common name	Family	No. of plants
Blackberry Chester	*Rosaceae*	25
Black raspberry	*Rosaceae*	29
Blueberry northsky	*Ericaceae*	26
Tayberry	*Rosaceae*	7
Honeysuckle emerald	*Caprifoliaceae*	23
Boysenberry	*Rosaceae*	16
Potential goldfinger	*Rosaceae*	25

The 100-m outlet channel discharging into Kingfisher Creek was constructed 10 m upstream of the Mabel Lake Road bridge. Three Newbury riffles were constructed within the outlet channel to provide resting ponds and create turbulent "noise" to attract adult spawning salmon. Large boulders were placed within the alcove at the mouth of the outlet channel to mimic a continuous stream bank and not interfere with the bridge abutment structures downstream. This design is intended to provide suitable access while maintaining the hydraulic conditions that break up back-eddies and retard sediment deposition into the channel outlet.

Livestock watering access and exclusion fences were established on both the Davison and Gerry properties to protect the riparian zone and channel. In areas with substantial disturbance to the soil by heavy equipment, 1,250 kg of erosion-control grass seed mixture (Table 1) was applied, and approximately 2,000 Pacific willow whips were planted. Along the lower 100 m of channel (adjacent to the Mabel Lake Road), 151 low-growing native shrubs were planted to provide shading while not impairing the sight lines of the property owners (Table 2).

10.5-km Site Upgrade

The works completed at the 10.5-km site on the FSR in 1997 included the creation of approximately 650 m of flowing (channel) and 6,000 m^3 of still (pond) water. This site is supplied by a groundwater infiltration system and utilizes an abandoned overflow channel within the Hunter's Creek floodplain. The habitat produced was augmented with LWD, riparian plantings, and the addition of spawning gravels to enhance the habitat quality. The works in 1998 included upgrading the trail system to restrict disturbance by visitors to the site. Several sections of the channel were also upgraded to improve fish access.

Dissolved oxygen (DO) readings were recorded using a YSI Model 51 meter. Measurements were taken along the channel from the spawning platform (groundwater discharge) and

at stations every 10 m downstream until concentrations had reached equilibrium (i.e., did not deviate by more than 1 mg/L for 3 consecutive stations). The data suggest that DO concentrations are low at the point of groundwater discharge (<5 mg/L at 15°C), but rise rapidly and reach equilibrium by the 25-m station (7–8 mg/L).

In-stream Works at the 12-km Site

The works completed at the 12-km site on the FSR included the stabilization of 2 eroding banks and the restoration of in-stream fish habitat throughout the B.C. Hydro right-of-way utility corridor crossing of Danforth Creek. As a result of the loss of all riparian vegetation and changes in the hydrologic regime from timber harvesting practices throughout the watershed, the (primarily sandy-silt) banks on both sides of the creek at this site exhibited signs of extreme lateral migration, undercutting, slumping, and sediment input. Using a combination of both traditional and bioengineering techniques, the site was stabilized, fish habitat was restored, and a riparian zone was established.

The slopes of undercutting banks were pulled back to approximately 1:3, leaving a trench at the toe of the slope equal to the depth of the thalweg[1]. (Note that the channel is not actively degrading within this reach.) Approximately 250 m^3 of 0.5- to 1.0-m angular rip-rap (sized according to Newbury and Gaboury 1993) was placed within the trench along the toe of the slope up to an elevation equal to that of the floodplain on the opposite bank. Three triangular LWD catchers (Slaney et al. 1997) were constructed to provide over-winter rearing habitat for juveniles and resting ponds for adult spawning coho. Each was constructed from fir or cedar logs with a minimum diameter of 0.5 m and length of 10 m. The apex was attached to a buried boulder (minimum diameter 1.5 m) using 7/16-inch galvanized wire rope and the Hilti epoxy system. The "butt" end was buried into the bank a minimum of 40% of the length. Within the centre of each structure, 1 large rootwad was anchored using wire rope and 2 "U" clamps. Forty-five modified brush layers were constructed above the rip-rap elevation using 8–10 whips, 1 cm in diameter by 1 m long, for each layer. Rebar (1 cm X 1 m) was used to secure the 15 cm X 1.5 m "riser" in place. The whips were buried approximately 80% into the soil, and all leaves and buds were stripped from the whips to prevent desiccation and encourage root growth. Finally, a grass seed mixture was applied to all disturbed soils to prevent surface erosion.

Newbury riffle structures were constructed with approximately 125 m^3 of 0.5- to 1.0-m angular rip-rap (sized according to Newbury and Gaboury 1993) placed across the channel and "keyed" into both banks. The riffle structure provides an upstream pool (approximately 50 m^2 with 1-m depth) and dissipates the flow's energy on the downstream (1:15) grade.

Several LWD-boulder complexes were constructed to protect eroding banks. Seven boulders (minimum 1.5 m diameter), 6 fir or cedar logs (minimum diameter of 0.5 m and length of 10 m), and 2 rootwads were anchored against the bank to provide direct protection to the flow and refugia habitat for fish during freshet flows. The Hilti system was used to anchor the materials in place.

In-stream Works at the 16-km Site

A B.C. Hydro access road and bridge crosses Danforth Creek at 16 km on the FSR. The riparian vegetation was relatively intact second growth (25–50 yr old), however the channel lacked suitable fish habitat prior to the initiation of this project. In situ gravel distribution indicates that this reach has the potential to constitute a primary spawning and rearing area.

In order to introduce hydraulic variability to this reach, 2 Newbury riffles and 3 triangular LWD catchers were constructed similar to the 12-km site. All works conducted within this site were completed using a Schaeff HS 41 "walking spider" excavator to minimize disturbance to the banks and riparian area.

At the upstream extent of this site, a naturally formed debris jam exists, creating several large pools and providing protection to a vulnerable bank. This structure was used as a template to design the constructed LWD-boulder features.

Midway downstream through this site is an access bridge. This structure was replaced in 1998 by B.C. Hydro to provide greater flood routing (more free-board and channel width). The orientation of the abutments was also modified to conform to the existing stream-flow lines.

At the downstream extent of this site, a debris jam was created during the 6 July 1997 flood event. This jam caused the channel to avulse to the east and create a new route. It is unclear whether the channel configuration will remain as it is or revert to its pre-1997 location. All works constructed in 1998 will function under both circumstances.

18.5-km Side-channel

Development and Floodplain Stabilization

Side-channel rearing development and floodplain restoration works were initiated within the B.C. Hydro right-of-way utility corridor crossing of Danforth Creek at 18.5 km on the FSR. As a result of the removal of the majority of riparian vegetation during corridor construction, the 6 July 1997 flood event, beaver dam-building activities, and changes in the hydrologic regime from timber harvesting practices further up in the watershed, the channel has been actively aggrading throughout this site, and the banks on both sides of the creek exhibit signs of instability and lateral migration.

Using the old-growth reach immediately upstream of this site as a template, structures were constructed to provide

[1] Thalweg: the path traced by the flow that follows the deepest part of the channel (Newbury and Gaboury 1993).

stability and protection to the banks and floodplain, and to encourage localized areas of channel-bed degradation. A "walking spider" excavator was employed within this reach due to the inaccessibility of the site by other equipment. Intact cedar LWD was scavenged from under the right-of-way and placed by overlapping and matrix building. Where ever required, wire rope and "deadmen" were used to secure structures in place.

It is anticipated that freshet flows will be concentrated between the LWD-reinforced banks and will encourage floodplain aggradation through sediment deposition. Unanchored LWD will be redistributed and form natural complexes. This reach also constitutes the greatest quality and quantity of gravel throughout the entire watershed and these in-stream structures will help to re-establish lost hydraulic variability and restore spawning areas to their pre-1997 flood productivity.

Three in-stream rearing ponds (each approximately 50 m^2 and 1 m deep) were constructed within the existing side channel and were loaded with LWD. It is anticipated that these ponds will remain ice-free during the winter due to groundwater temperature influences and will provide critical over-wintering coho salmon habitat. They will also act as flood-flow refugia during freshet.

PRODUCTION ESTIMATES

Biostandards for fish habitat enhancement projects were used to estimate smolt/juvenile production for coho salmon, and resident and adfluvial rainbow trout in the Kingfisher Creek watershed (Koning and Keeley 1997). Rainbow trout estimates were derived from coastal steelhead salmon levels (P. Slaney, pers. comm.). These estimates represent potential population target values and are influenced by adult escapement numbers and future changes within the watershed.

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Response of Fish Populations to Weir Construction in Arden Creek

Tracy A. Michalski
Fisheries Section
British Columbia Ministry of Environment, Lands and Parks
2080 Labieux Road, Nanaimo, BC, V9T 6J9, Canada
Tracy.Michalski@gems4.gov.bc.ca

George E. Reid
Fisheries Section
British Columbia Ministry of Environment, Lands and Parks
2080 Labieux Road, Nanaimo, BC, V9T 6J9, Canada

ABSTRACT

In 1996, the Courtney Fish and Game Protective Association constructed 2 wooden weirs and 1 rock weir in a channelized portion of Arden Creek, Vancouver Island, B.C. The Association added an additional wooden weir in 1997. Volunteers from the Association, students from Puntledge Park and Lake Trail schools, and Urban Salmon Habitat Program staff conducted fish assessments at weir (treated) and control (untreated) sites in 1996 and 1997. We found more fish at treated sites in both years. In 1996, 66% of the fish sampled were associated with weirs, and in 1997, 80%. The density of fish at treated sites was 0.67 fish/m^2, compared to 0.35 fish/m^2 at control sites in 1996, and 0.29 fish/m^2 at treated sites compared to 0.07 fish/m^2 at control sites in 1997. We estimated the biomass to be 3.32 g/m^2 at treated sites and 2.4 g/m^2 at control sites in 1996. In 1997, we estimated the biomass at 3.87 g/m^2 at treated sites and 0.35g/m^2 at control sites. When we compared the population and biomass statistics between weir types, we found that wooden weirs produced greater absolute numbers, higher densities, and higher biomass. The mean length of sampled coho increased significantly between 1996 and 1997, due mainly to density dependent factors. We make a number of recommendations for future work on Arden Creek.

Key words: coho, cutthroat, *Oncorhynchus clarki*, *Oncorhynchus kisutch*, habitat restoration, weirs.

Arden Creek is a small, urban creek located in the municipality of Courtenay on Vancouver Island, B.C. (Fig. 1). The creek has undergone extensive channelization, resulting in a loss of habitat complexity. Approximately 1 km in length, Arden Creek flows unaltered for approximately 200 m through a protected, wooded wetland before entering School District 71 property. The remaining 800-m section of the creek has been straightened and follows the boundary along Puntledge Park Elementary School. The creek supports both coho (*Oncorhynchus kisutch*) and cutthroat trout (*O. clarki*). It has historically suffered from low summer flow (Law 1994).

Raven River Habitat Services conducted a survey of habitat enhancement opportunities in the Comox Valley in 1992 and recommended augmenting the flow and increasing the habitat complexity in Arden Creek (Baldwin 1992). The federal Department of Fisheries and Oceans (DFO) and the British Columbia Ministry of Environment, Lands and Parks (MELP) cooperated to address these recommendations in 1993 and constructed a flow diversion structure to channel 0.03 m^3/sec of water from Morrison Creek into Arden Creek. Complexes of large woody debris (LWD) were cabled into several locations upstream of the channelized portion of the creek.

In 1996, The Courtenay Fish and Game Protective Association applied for funding from the Urban Salmon Habitat Program (USHP) to enhance the complexity of Arden Creek habitat where it flows through School District 71 property. The Association successfully obtained this funding, and in 1996 and 1997 constructed 3 wooden weirs and 1 rock weir in the channellized area of the creek. The objective of the project was to create additional habitat for rearing fish. Weirs encourage downstream scour and create dammed pools upstream (Murphy 1995, Koning and Keeley 1997). Constructing these structures would restore the pool habitat critical to the survival of both coho salmon and cutthroat trout.

Volunteers from the Courtenay Fish and Game Protective Association, students from Puntledge Park and Lake Trail schools, and USHP staff conducted fish assessments at weir and untreated (control) sites in 1996 and 1997. Our objective was to monitor changes to fish populations as a result of the installation of the weirs. In this paper we present the results of these assessments and make suggestions for future work on Arden Creek.

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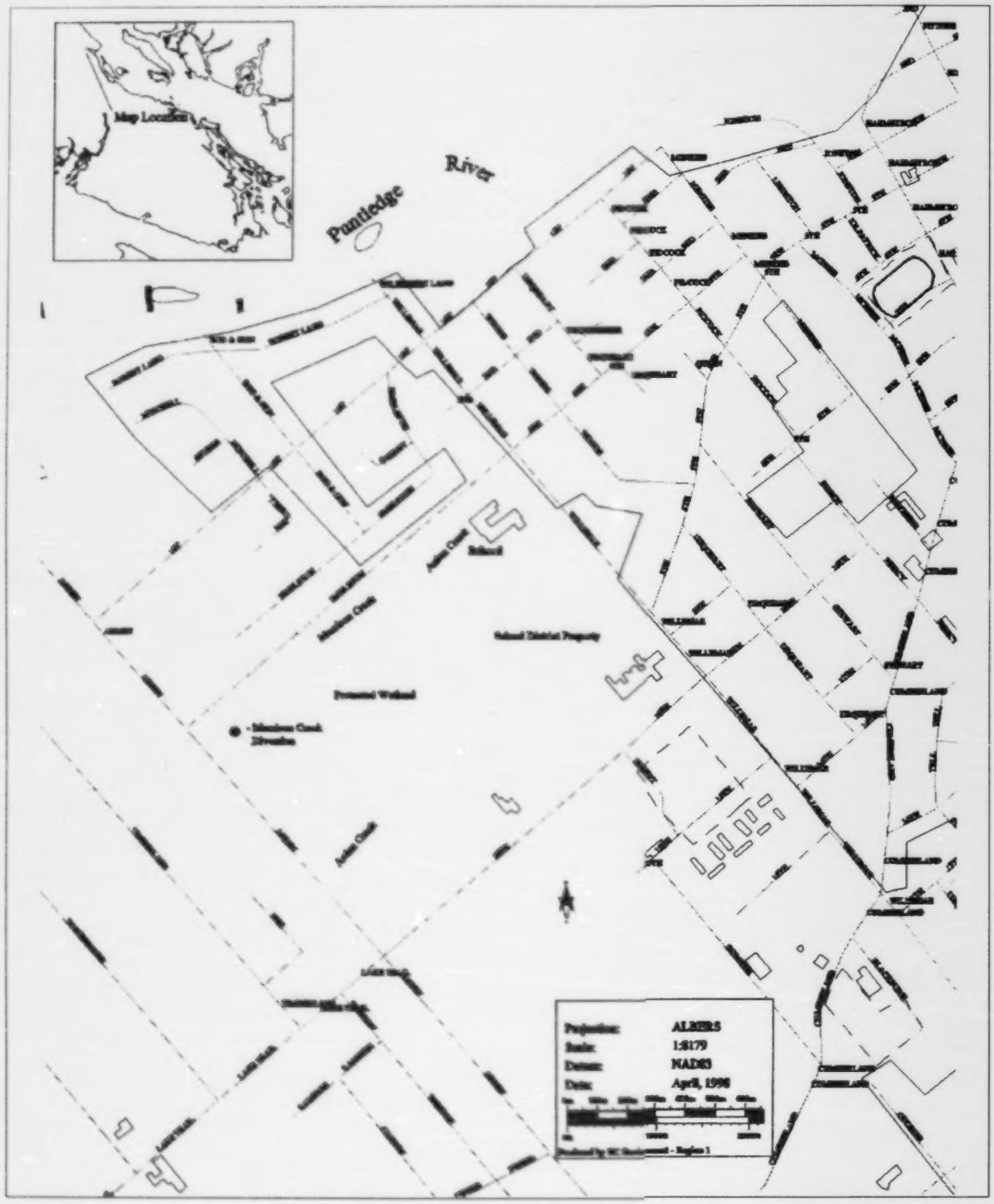


Figure 1. Location of Arden Creek, Courtenay, Vancouver Island, B.C.

METHODS

In October of 1996 and 1997 we sampled fish at the weirs and control sites of Arden Creek, with the assistance of the Courtenay Fish and Game Protective Association and students from Puntledge Park and Lake Trail schools. We used the Two-Pass method to sample the fish (Resource Inventory Committee 1993). We netted off areas above and below the sample sites to prevent fish migration and removed fish using an electroshocker. Once we had sampled the fish, they were anaesthetized, enumerated, identified to species, weighed, and measured. All fish were then returned to their capture location.

We analyzed the data by comparing the total abundance, density (fish/m^2), and biomass (g/m^2) of sampled fish. We also examined species length frequencies by categorizing fish lengths into 10-mm classes and comparing frequencies by year and site.

RESULTS

Coho and Cutthroat Abundance and Population Density

The number of fish sampled in Arden Creek in 1996 and 1997 totalled 130, of which 85% were coho (Table 1). We sampled 85 coho in 1996 and 25 in 1997. The difference between the 2 years was significant ($P = 0.03$). Cutthroat abundance displayed a similar pattern, but their numbers were proportionately reduced. We sampled 17 cutthroat in 1996 and 3 in 1997.

We found more fish at treated sites than at control sites in both 1996 and 1997 (Table 1). In 1996, 67 fish were taken at the treated sites, compared with 35 at the control sites. Of the 67 fish sampled, 55 were coho and 12 were cutthroat, compared to 30 coho and 5 cutthroat from the control sites that year. We only sampled 28 salmonids in 1997. Twenty-five of these were coho, 80% of which were found at treated sites. Only 3 cutthroat were sampled in 1997; 2 were at treated sites.

We compared population densities at each site by relating the number of fish sampled to the area sampled (Table 2) and found a significant decrease in the density of coho between the two years ($P = 0.07$). We then examined the difference in density between treated and control sites for both years. In 1996, the population density at treated sites was 0.67 fish/m^2, 65% higher than the 0.35 fish/m^2 sampled at the control sites. In 1997, we sampled 0.29 fish/m^2 at the treated sites compared with 0.07 fish/m^2 at the control sites. Treated sites produced higher densities for individual species in both 1996 and 1997. The density estimates for coho between 1996 and 1997 at the treated sites were 0.55 and 0.12 fish/m^2 respectively, compared to 0.3 and 0.06 fish/m^2 at the control sites. The pattern for cutthroat was similar.

Table 1. Coho salmon and cutthroat trout abundance sampled at treated and control sites in Arden Creek in 1996 and 1997.

	1996		1997		Total
	Treated	Control	Treated	Control	
Area sampled (m^2)	100	100	205	85	n/a
Coho	55	30	20	5	110
Cutthroat	12	5	2	1	20
Total	67	35	22	6	130

Coho and Cutthroat Biomass

We did not find any difference in the total amount of biomass produced between 1996 and 1997. In 1996, treated sites produced 3.32 g/m^2, control sites 2.4 g/m^2 (Table 2). In 1997, the biomass at treated sites was 3.87 g/m^2, 91% higher than the 0.35 g/m^2 sampled at control sites. Coho biomass declined significantly between the 2 years, while cutthroat biomass increased significantly in that same period, accounting for the small total difference between years ($P \leq 0.001$).

When we combined biomass for both species for both years, treated sites produced 7.25 g/m^2, and controls site 2.75 g/m^2. The total biomass of coho captured at treated sites in 1996 was 2.86 g/m^2, significantly higher than that at control sites ($P = 0.004$). The biomass of cutthroat was 63% higher at the treated sites in 1996. In 1997, cutthroat biomass was higher at treated sites; however, we only captured 3 fish, 1 of which was a 2+ fish (parr) which accounted for most of the total biomass.

Table 2. Coho salmon and cutthroat trout densities (fish/m^2) and biomass (g/m^2) at treated and control sites in Arden Creek in 1996 and 1997.

	1996				1997			
	Treated (n = 67)		Control (n = 35)		Treated (n = 22)		Control (n = 6)	
	Density	Biomass	Density	Biomass	Density	Biomass	Density	Biomass
Coho	0.55	2.86	0.30	2.1	0.12	2.14	0.06	0.29
Cutthroat	0.12	0.52	0.05	0.30	0.15	1.73	0.01	0.06
Total	0.67	3.32	0.35	2.40	0.29	3.87	0.07	0.35

Coho and Cutthroat Lengths

The mean length of coho increased significantly between years, from 59.8 mm in 1996 to 71.52 mm in 1997 ($P \leq 0.001$; Fig. 2). In 1996, coho ranged between 44 and 90 mm, with a modal length of 50–59 mm. In 1997, the length of coho ranged between 55 and 90 mm and the modal length had increased to 60–69 mm.

The cutthroat sampled in 1996 ranged between 40 and 87 mm. The modal length of these fish was 50–59 mm. This compares to 1997, when the length ranged from 72 to 168 mm. The 1996 samples were made up entirely of 0+ fish (fry), while in 1997, the 3 fish sampled represented 3 different age classes.

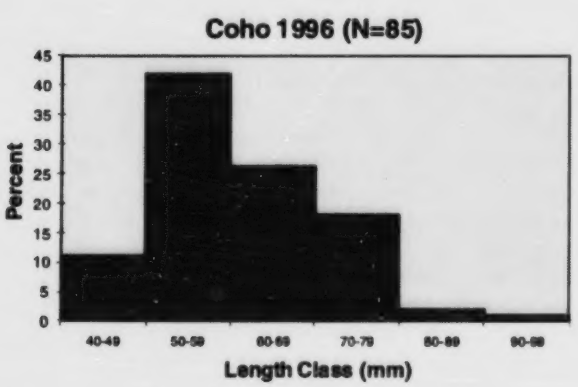
We found larger coho at control sites in both 1996 and 1997 (Fig. 3). In 1996, the mean length of coho at control sites was 62.9 mm; this was almost 5 mm larger than the coho sampled at treated sites. The mean length of coho sampled at control sites in 1997 was 75.4 mm compared to 70.5 mm at treated sites. In 1996, the cutthroat sampled at control sites averaged 61 mm, while the average length at treated sites was 55.5 mm (Fig. 4). We did not compare lengths in 1997.

Fish Abundance, Density, Biomass, and Length at Wooden vs. Rock Weirs

We sampled 67 salmonids at the 2 treated sites in 1996 (Table 3). Of these, 55 were coho, 69% of which were sampled at the wooden weir, and 12 were cutthroat, 83% of which were captured at the wooden weir.

Density estimates were also consistently higher at the wooden weir in 1996 (Table 3). The total density sampled at both rock and wooden weirs was 1.58 fish/m^2. Of that, 1.23 fish/m^2 were captured at the wooden weir and 0.35 fish/m^2 at the rock weir. Coho density at the wooden weir was 0.87 fish/m^2, more than twice the estimated 0.35 fish/m^2 found at the rock weir. Cutthroat density at the wooden weir was almost 10 times the 0.04 fish/m^2 sampled at the rock weir.

Biomass estimates were also higher at the wooden weir in 1996 (Table 3). The total biomass for all fish was 2.2 g/m^2 at the wooden weir and 1.1 g/m^2 at the rock weir. Wooden weirs produced an estimated 1.9 g/m^2 of coho, compared to 0.96 g/m^2 for the rock weir. The cutthroat biomass of 0.37 g/m^2 at the wooden weir was more than 2 times that at the rock weir.



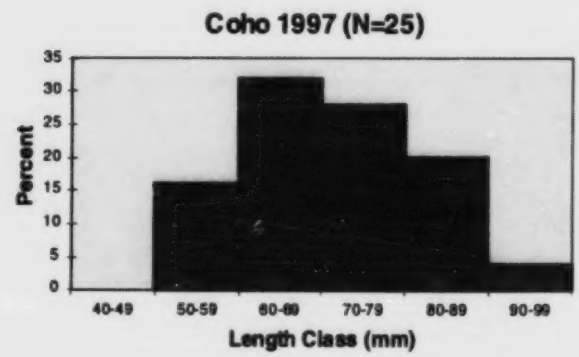


Figure 2. Comparison of length classes of coho salmon sampled in Arden Creek in 1996 and 1997.

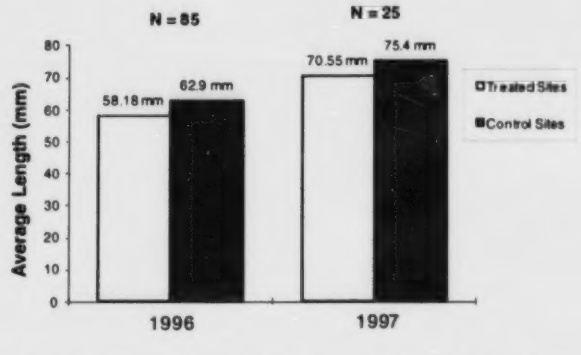


Figure 3. Average lengths of coho sampled at treated and control sites in Arden Creek in 1996 and 1997.

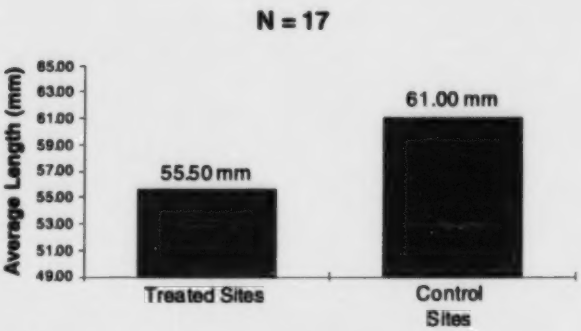


Figure 4. Average lengths of cutthroat sampled at treated and control sites in Arden Creek in 1996.

Table 3. Comparison of total abundance, population, and biomass estimates at wooden and rock weir sites in 1996 and 1997.

	1996		1997	
	Wooden weir (n = 1)	Rock weir (n = 1)	Wooden weirs (n = 3)	Rock weir (n = 1)
Sample area (m^2)	50	50	111	94
Coho				
No. sampled	38	17	11	9
Population estimate (fish/m^2)	0.87	0.35	0.14[a]	0.26
Biomass (g/m^2)	1.9	0.96	0.59[a]	0.38
Cutthroat				
No. sampled	10	2	2	0
Population estimate (fish/m^2)	0.36	0.04	0.02[a]	0
Biomass (g/m^2)	0.37	0.15	0.51[a]	0
All fish				
No. sampled	48	19	13	9
Population estimate (fish/m^2)	1.23	0.35	0.16[a]	0.26
Biomass (g/m^2)	2.2	1.1	1.1[a]	0.38

[a] Indicates average for all sites sampled.

We continued to find a greater absolute abundance at the wooden weirs in 1997 despite the fact that we found higher densities at the rock weir. We sampled 11 fish at the wooden weirs that year, compared to 9 at the rock weir. In contrast, the density of fish at the rock weir was 0.26 fish/m^2, compared to 0.16 fish/m^2 at the wooden weirs (Table 3). However, the biomass estimate for all salmonids at the wooden weirs was 1.1 g/m^2, significantly higher than the 0.38 g/m^2 estimated for the rock weir ($P = 0.016$).

The average length of coho sampled at the rock weir in 1996 was 62.59 mm, compared to 56.2 mm at the wooden weir. This difference was significant between the 2 weir types ($P = 0.025$). The pattern was the same for cutthroat that year, with larger fish being sampled at the rock weir. In 1997, the pattern was reversed for coho, with larger fish sampled at the wooden weir. No length comparison was done for cutthroat sampled in 1997.

DISCUSSION

Coho and Cutthroat Abundance, Population Density, and Biomass

We found a decrease in abundance, population density, and biomass of coho, and a decrease in abundance and density of cutthroat trout in Arden Creek between 1996 and 1997. Decreases in fish population size and biomass between years are caused by changes in escapements, rearing densities, temperature, food supply, and/or changes in the amount of available habitat. Escapements of coho into the Puntledge River system in 1996 were below average, but strong enough to seed the available habitat (B. Allen, DFO, June 1998, pers. comm.), therefore, we do not attribute the decreases we saw to a decrease in escapement. The average temperatures were higher between April and October in 1997 (Environment Canada, 1998, unpubl. data from Courtenay/Meadowbrook AES weather station), but higher temperatures would have a positive effect on the population size and biomass. We do not believe that food supply was limited. The mean lengths of the coho we sampled in both 1996 and 1997 were consistent with the findings of Rounsefell and Kelez (1940), who reported that with moderate water temperatures and an abundant food supply, coho fry will grow from 30 mm at emergence in March to 60–70 mm in September. In the summer of 1997, however, volunteers from the Courtenay Fish and Game Protective Association noted that Arden Creek was dry in several locations. The decrease in flow was due to accumulated debris and bedload in the Morrison Creek diversion. We believe this limited the amount of habitat available in Arden Creek. We attribute the decrease in fish populations in Arden Creek between years to this.

In 1997, we found a significant increase in the size of coho and cutthroat sampled in Arden Creek. We attribute this to a number of factors including: fewer fish in 1997 competing for food and space; and warmer temperatures between April and October 1997, resulting in earlier emergence and a longer growing season. In addition, the decrease in flow resulted in decreased flushing rates in Arden Creek. This would lead to pooling and, ultimately, increased temperatures in the pooled water, which would have a positive effect on fish growth provided the temperatures remained <20°C.

Fish Abundance and Length at Treated vs. Control Sites

We found more fish as well as higher densities and biomass at treated sites in both 1996 and 1997. We attribute this to the

creation of deeper pool habitats at the weir sites. Although coho are found in both pool and riffle habitat, they are best adapted to holding in pools (Hartman 1965). Researchers have found that the highest absolute and biomass densities of coho are correlated with depths of between 10 and 40 cm and that coho prefer depths of 30 cm or more (Glova and Masson 1976, Nickelsen 1975). The weirs in Arden Creek created upstream pool depths of between 26 and 36 cm, while the control areas had an average depth of only 13.3 cm.

We found larger fish at control sites in both 1996 and 1997. Salmonids in streams defend territories from small, post-emergent juveniles until they become smolts or sexually mature. These fish increase the area they defend to meet increasing energetic requirements (Keeley and Slaney 1996). This results in decreasing population density as average body size within a cohort increases (Grant and Kramer 1990). In small or very small streams, a conflict of overlapping territories can inhibit growth, encourage emigration, or both (Marshall and Britton 1980). We postulate that as the number of fish increased at treated sites, fish emigrated to control sites, where there was more space. The lower densities of fish competing for territory and resources in control sites resulted in greater growth at these sites.

Fish Abundance and Length at Wooden vs. Rock Weirs

We also found more fish and higher biomass at wooden weirs in both 1996 and 1997. Both the rock and wooden weirs created similar pool habitat; the only difference is the location of the weirs. The rock weir is located at a stream crossing site used by students and members of the public. People walk across the top of the weir and dislodge the rocks. This traffic also causes bank sloughing, resulting in inputs of sediment into the stream.

The effects of suspended sediment on rearing salmonids are twofold, affecting both the water column and the streambed (Griffith 1980). Light penetration to the streambed is diminished with increasing sedimentation, which, in turn, limits the primary production and availability of food. Thompson (1972) reported decreasing potential productivity with decreasing substrate particle size, silt being the least productive. Furthermore, salmonids are sight-feeders and Bachman (1958) observed the cessation of feeding by cutthroat at turbidity levels of 25 ppm. Herbert and Merkens (1961) report that sediment levels as low as 90 ppm can adversely affect the survival of trout. We conclude that the inputs of sediment and disturbance by foot traffic are likely responsible for the lower number of fish at the rock weir site.

MANAGEMENT IMPLICATIONS

The construction of weirs in Arden Creek has had a positive effect on the fish production of this small urban creek. We suggest, therefore, that the creek continue to be a focus for restoration activities by the club.

Arden Creek was dry in several locations due to accumulated debris and bedload in the Morrison Creek diversion in the summer of 1997. This resulted in decreased rearing habitat in Arden Creek, which contributed to the decrease in the number of fish in the creek that year. DFO and MELP have now developed both short- and long-term prescriptions to address this situation. In the short term, constructing a weir in Morrison Creek will create an upstream pool and back water into the diversion and then into Arden Creek (R. Doucet, DFO, 1998, pers. comm.). The longer-term solution involves burying a pipe into the bed of Morrison Creek to ensure a permanent flow of water into Arden Creek.

We found both rock and wooden weirs increased the number of fish in Arden Creek. Therefore, we recommend construction of additional weirs in the creek and suggest that the Courtenay Fish and Game Protective Association commission a study to determine appropriate locations for these structures. More natural-looking structures, such as Newbury weirs should be considered (Newbury et. al. 1997). These weirs are constructed out of boulders and are spaced to create continuous pool and riffle habitat.

Large woody debris provides important physical and biological functions in the wide variety of habitats used by all species of Pacific salmon (Cederholm et. al. 1997). In their study of the summer microdistribution of coho underyearlings in the Big Qualicum River on Vancouver Island, Lister and Genoe (1970) reported that fry were in close association with bank cover, in back eddies, log accumulations, and rootwads. Even in the deeper, faster, mid-stream areas, juvenile coho still remain in close proximity to cover. Ward and Slaney (1997) compared differences in fish populations at sites with boulder deflectors, with and without LWD, in the Keogh River on Vancouver Island. Deflectors with LWD carried about twice as many juvenile steelhead parr and young-of-year coho.

We recommend that LWD and rootwads be secured in Arden Creek, particularly along the channelled portion of the creek on School District 71 property and in the upstream pools created by the weirs. Not only would the LWD complexes provide biological benefits, they would also trap sediment. Cederholm et. al (1997) found that more than half of the total sediment stored in first to third order streams is retained by organic matter such as LWD.

Sediment at the rock weir site is having a localized impact on fish and fish habitat at this location. There are numerous similar crossing sites throughout the School District 71 property and within the wooded wetland upstream. In his investigation, Baldwin (1993) noted that the substrate of much of Arden Creek was silt and organic muck over a layer of sand and gravel, and he speculated that the silt was probably coming from bank erosion caused by foot traffic. He suggested the construction of additional bridges and bank stabilization with riparian plants. We recommend this be done. In addition to the

impacts from people crossing the creek, upper Arden Creek has been subject to impacts from urban development. It is likely that some of the sediment also comes from these areas.

Once sediment inputs are eliminated or minimized, there are a number of ways to dislodge the existing sediment. One is scarficiation, which involves excavating the streambed and exposing the substrates to the flushing effect of the stream. Another technique is hydraulic flushing, which uses high pressure jets of water and air to force the fines into the flow of the stream (Adams and Whyte 1990). Constructing instream deflectors will remove sediment and assist in the restoration of sediment-impacted channels by creating pools and increasing cover by narrowing and deepening the stream (Koning and Keeley 1997). We recommend that an assessment be undertaken to determine the sediment sources and to develop appropriate prescriptions to eliminate the inputs. We believe that instream structures combined with weir construction would create more habitat and increase the movement of sediment.

Finally, we recommend continued monitoring of fish populations in Arden Creek. Monitoring provides valuable information regarding changes to fish populations and will expand the knowledge base about changes to fish populations as a result of restoration activities. In addition to monitoring fish, we recommend ongoing monitoring of all structures placed in the creek to ensure they are working appropriately and to determine whether any changes or adjustments should be made. Monitoring should include examining changes to the stream to ensure that the structures are having long-term, positive effects on restoring this small but valuable urban creek.

ACKNOWLEDGEMENTS

We sincerely thank the members of the Courtenay Fish and Game Protective Association for their dedication to the restoration and protection of Arden Creek. Their tireless commitment and long hours installing weirs in the summers of 1996 and 1997 not only helped to increase the fish production of this small creek, but also led to valuable information about the changes to fish populations in restored urban streams. We also thank the students from Lake Trail Secondary and Puntledge Park Elementary schools for their assistance in collecting some of the fish data. Our thanks to L. Carswell and C. Thirkill for their diligence during the field sampling, and G. Carswell for her enduring volunteer commitment to fish on Vancouver Island.

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Restoration and Species at Risk Recovery: The Urban Salmon Habitat Experience

Patty Thomas[1]

Restoration of Natural Systems Program, University of Victoria, Environmental Studies
P.O. Box 1700, Victoria, BC, V8W 2Y2, Canada

ABSTRACT

A partnership of educational institutions evaluated and monitored selected projects supported by the Urban Salmon Habitat Program (USHP) on the east side of Vancouver Island, B.C. Students collected data according to USHP standards, and presented their findings at a colloquium in March 1998. Our results indicate that project participants misunderstood many restoration concepts. This led to much unusable data. As far as community/educational restoration projects go, a clear understanding of concepts and protocols is necessary for success, and hence recovery of endangered species and their habitats.

Key words: coho, monitoring, *Oncorhynchus kisutch*, restoration, stewardship, urban.

Slaney et al. (1996) found that of the 393 total salmon stocks[2] identified in the Georgia Strait/Vancouver Island region, 26 were extinct, 31 were at high risk, 10 were at moderate risk, and 27 were of special concern. There are a high number of coho (*Oncorhynchus kisutch*) stocks in each of these risk categories (Slaney et al. 1996) because urbanization has greatly impacted this species' habitat. Notably, since the 1950s, the number of streams that supported spawning coho in the Georgia Basin[3] has declined from about 100 to around 20 (Province of British Columbia 1995).

Many volunteers and students are involved in various "species and habitats at risk" initiatives (e.g., the University of Wisconsin Arboretum, the Chicago Wilderness Project, the Mattole Watershed Project in Petrolia, CA). Similarly, the Urban Salmon Habitat Program (USHP) relies in part on community stewardship groups and educational institutions to achieve their goals within the rapidly urbanizing Georgia Basin.

The USHP funds nonprofit organizations that undertake 1 or several of the following activities:

1. rehabilitation and restoration of urban salmon streams;
2. community-based watershed planning;
3. assessment and mapping of salmon habitat;
4. public awareness and education;
5. monitoring and evaluation projects; and
6. land-owner contact.

The USHP provides technical advice and training to support these nonprofit groups in their various efforts to improve fish habitat. USHP complements other stewardship initiatives such as the Department of Fisheries and Oceans' Streamkeepers Program (Province of British Columbia 1995).

Our problem statement was: "The assessment, restoration and public awareness/education projects funded under USHP need to be monitored to determine the success of individual projects and the USHP as a whole. There is currently no formal mechanism for measuring the performance of the proposals funded under the USHP."

The objectives of our project were:

1. to monitor the USHP site assessment and mapping projects;
2. to monitor the USHP restoration projects;
3. to monitor the USHP education and public awareness projects; and
4. to present the findings of the monitoring program to students and stewardship groups in a conference, and to prepare a proceedings from the conference.

METHODS

We began the project by setting up a core group consisting of: G. Reid and T. Michalski from the British Columbia Ministry of Environment, Lands and Parks (MELP); R. Hebda, R. McDougall, and P. Thomas from the University of Victoria (UVIC); J. Merriman and C. Chestnut from Malaspina University-College; M. Corry and P. Marshall from

[1] Present address: Fisheries Renewal British Columbia, 405 - 960 Quayside Drive, New Westminster, BC V3M 6G2, Canada
Patty.Thomas@gems3.gov.bc.ca

[2] A stock is a partially isolated group of 1 salmon species that spawns in a specific stream at a specific time.

[3] The Georgia Basin consists of the Lower Mainland (from Hope to Vancouver), the Sechelt peninsula, and the east coast of Vancouver Island (from Campbell River to Victoria) in the province of British Columbia.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Table 1. Evaluation of student monitoring data collected according to USHP protocol relative to USHP monitoring standards (±5% of the calculated values).

Parameter	No. of data sets collected following USHP protocol	No. of data sets meeting USHP standards	Percentage of data points meeting USHP standards
Percent pool area	2	0	0
LWD(large woody debris) /bankfull channel width	5	Ayum Creek, Reach 1	20
Erosion sites	2	0	0
Obstructions	5	Ayum Creek, Reach 2 Ayum Creek, Reach 3	40
Altered stream sites	5	Ayum Creek, Reach 2 Ayum Creek, Reach 3	40
% wetted area (wetted area /total area)	5	0	0
pH	1	Ayum Creek, Reach 1	100
Location in meters of reach breaks	1	0	0
No. of reach breaks	2	0	0
Land use	2	0	0
Riparian slope	2	0	0
TOTAL	32	6	19

Camosun College; and E. Britton from North Island College. This core group met several times to discuss the project. Since UVIC was coordinating the project, we held additional meetings with MELP staff.

To explain the program and the projects, USHP staff conducted introductory workshops and demonstrations for the students. At these workshops 2 documents, *Assessment and Mapping Procedures for Vancouver Island* (Michalski et al. 1997) and *Standards and Procedures for Assessing Urban Salmon Habitat Projects on Vancouver Island* (Michalski and Reid 1997), were distributed to the students and instructors. The students also received a computer disk containing an EXCEL-based spreadsheet program (described below). In addition, USHP staff answered students' questions and made field visits to some project sites as needed (e.g., to Thatcher Creek to assist with electrofishing procedures).

The students received previous reports and stream monitoring data collected by stewardship groups and were taught how to collect stream monitoring data according to USHP protocols. They then collected data at a stream and compared their data to the data previously collected by stewardship groups. Acceptable values were ±5% or ±15% of the values reported by the stewardship groups, depending on the parameter measured (Tables 1 and 2).

We analyzed the results of our project in several ways. We compared the students' methodology to the procedures outlined in Michalski et al. (1997) and Michalski and Reid (1997). We analyzed the students' data to see if they met the standards of ±5% or ±15% presented in Michalski and Reid (1997). We held a colloquium on 21 March 1998 at Malaspina University-College for students, instructors, and stewardship groups. At this gathering, students presented their reports, and T. Michalski and R. Hebda talked about the USHP. Comments about the project were solicited from the audience and incorporated into this report. Additional comments about the project were obtained from telephone interviews with stewardship groups.

We need to highlight a few general points about the USHP assessment and monitoring procedures because these influence the interpretation of the results. The USHP assessment approach (Michalski et al. 1997) was adapted from other assessment procedures (e.g., Johnston and Slaney 1999). The EXCEL-based spreadsheet program automatically summarizes and compares data to standards for undisturbed watersheds. The USHP approach suggests that the habitat, fisheries, and riparian assessments be done together. If this cannot be done, the USHP assessment manual recommends doing the habitat and riparian assessments first (Michalski et al. 1997).

We did not monitor or evaluate public education programs because, by mutual agreement, it was felt that this type of evaluation was beyond the training of students presently involved in the program. We are looking into funding a graduate student to undertake this type of evaluation.

RESULTS

Tables 1 and 2 summarize the results of the students' work. In them we only included data that were collected according to USHP protocol outlined in Michalski and Reid (1997).

Table 2. Evaluation of student monitoring data collected according to USHP protocol relative to USHP monitoring standards (±15% of the measured values).

Parameter	No. of data sets collected following USHP protocol	No. of data sets meeting USHP standards	Percentage of data points meeting USHP standards
% cover in pools	3	Ayum Creek, Reach 1 Ayum Creek, Reach 3	67
Livestock length	2	Upper Nile, Reach 2 Lower Nile, Reach 1	100
Average % boulder cover	4	0	0
% crown cover	5	Ayum Creek, Reach 1 Upper Nile, Reach 2	40
Stability	2	0	0
TOTAL	16	6	38

Thus, we generally omitted data that were collected at the wrong points in the stream (e.g., data not collected at the 0- and 200-m marks) and in the wrong way (e.g., not according to the protocol). In addition, we omitted some data because we were not clear what they represented (e.g., unclear acronyms and locations, and incomplete calculations). For specific details, please refer to a discussion of the individual case studies found in Thomas and Hebda (1998).

When we compared the 48 data sets collected by the students according to the USHP protocol to the stewardship groups' data, only 12 (25%) met the USHP standard. Even this value is misleading, because if we omit data that met the standards because of a "0" value (i.e., livestock length, obstructions, and altered stream sites), only 7 (15%) of the students' findings meet the USHP standards.

DISCUSSION AND RECOMMENDATIONS

The relatively poor agreement between data collected by stewardship groups and students, and the audience's comments at the colloquium raise several questions, the main one being, why are there such apparent differences? These differences may arise for 3 basic reasons:

1. Conceptual issues (i.e., a lack of understanding of the restoration process and the differences between, and the roles of, evaluation and monitoring).
2. Organizational factors that make taking comparable measurements difficult.
3. Problems with the measurement tool.

Conceptual Issues

We recognized that not all of the projects' participants understood what our terms meant and what the project was about. Thus, some students did not follow the monitoring protocol and produced unusable data. In addition, some students seemed to want to have their data "match" that of stewardship groups. Although this scenario is understandable, it yields questionable results.

In our second year of the project, we plan to hold informal meetings between students and stewardship groups at the start of the project to review the following:

- the basic steps of restoration;
- the role and differences of baseline assessment, monitoring, and evaluation in restoration projects; and
- the importance of evaluating restoration projects.

Organizational Factors

At the outset of this project we knew there was a potential problem with educational institutions' schedules and the sampling protocols (Michalski et al. 1997). For results to be comparable, measurements need to be made generally at the same time. For various reasons, especially course scheduling, this was not possible. This was 1 of the main reasons for the poor agreement between students' and stewardship groups' data (Tables 1 and 2).

We believe that we need to "fine-tune" the monitoring process for each educational institution. Next year, we plan to have different institutions collect different types of data, depending on the time of year students are able to do their work.

Problems with the Measurement Tool

There are 3 major factors that can cause variation in data:

1. The measurement tool works poorly and gives inconsistent readings (e.g., an uncalibrated pH meter, different interpretation of reach breaks).
2. The humans making the measurements do so inconsistently (e.g., riparian cover is estimated differently by different people).
3. The phenomenon being measured varies markedly (natural variation).

The first 2 factors can be controlled and corrected for by choosing and testing the appropriate tools (methods) and by training the measurers, respectively. Both of these factors will be addressed in next year's project. The third factor, the natural variation, cannot be corrected for; it can only be described and statistically accounted for.

The degree of natural variation present in an ecosystem can only be understood once a sufficient number of

measurements have been made using standard approaches (i.e., baseline data). The standards we used to compare the students' and stewardship groups' data did not adequately account for natural variation. Thus, in some cases, though the data were obtained according to the protocol, they did not meet the ±5% or ±15% standards because these standards were too narrow to account for natural variation. We recognize that this approach is clearly problematic, from both an ecological and mathematical perspective.

We will continue to work with MELP to come up with appropriate and practical standards for monitoring this program. We will continue to collect objective and subjective data from assessment and restoration projects and analyze the results carefully.

CONCLUSIONS

Our USIIP project had 3 types of objectives: evaluation, data gathering, and education. These objectives were linked to the broader goal of improving and validating the USIIP program.

The evaluation objective had 2 aspects. First, we specifically evaluated the USIIP's methods and standards. To be effective, the USIIP methods have to be: scientifically sound; useful for the purpose intended; practical in terms of the skills of the users; and unambiguous. Our analysis reveals that these criteria were not always met.

Second, we evaluated the stewardship group projects. Here we were concerned with whether the groups did what they said they would do and if they did it in an effective way. From the analysis of the student reports, it is clear that stewardship groups did indeed do what they said that they would do. Whether all of their work was effective or not is unclear. As we have discussed, the main issue related to this point is use of appropriate data gathering methods and standards.

We also fell short of our objective of data gathering. Many students did not gather their data according to USIIP standards. However, their data may be useful for baseline purposes.

Although our project did not go exactly as planned, we certainly met our educational objectives. We had 20 students involved in data collection and approximately 75 students and stewardship group members attended our colloquium. Students expressed appreciation for the learning opportunities the USIIP conferred on them.

In conclusion, our first year involved many students and stewardship groups, raised the profile of the USIIP, and tested the USIIP standards approach. We learned much about running a complex restoration project. The knowledge and experience gained will serve us well in redesigning evaluation, monitoring, and basic data gathering in the years to come. We also believe our experiences will help others who are contemplating a similar undertaking. With this improved knowledge we hope that the work of the stewardship groups will be more effective in bringing back at risk salmonids to urban areas.

ACKNOWLEDGEMENTS

First, we thank G. Reid and T. Michalski from the B.C. Ministry of Environment, Lands and Parks for their ongoing commitment to this project and their support of our Restoration of Natural Systems Program at the University of Victoria. We deeply appreciate the hours they have spent explaining the program to us and reviewing our report. We thank P. West, Director of the School of Environmental Studies, University of Victoria, for reviewing our report.

Second we thank the instructors involved in this project: J. Merriman and C. Chestnut from Malaspina University-College in Nanaimo; M. Corry and P. Marshall from Camosun College in Victoria; and E. Britton from North Island College in Campbell River.

Third, we greatly appreciate the students' hard work, which provided us with data to evaluate. We praise the students' honest reports of their successes and failures. It is this type of feedback that helps us and others improve on the vital work of restoring fish and fish habitat.

Finally, we thank those remarkable stewardship groups, the life blood of the USIIP. Their many hours of selfless work is a testimony to the drive of humans to live within the environment.

We thank the USIIP for supporting our work on this project.

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Effects of Habitat Fragmentation on Amphibians: What Do We Know and Where Do We Go From Here?

Elke Wind
Centre for Applied Conservation Biology
3004 - 2424 Main Mall, University of British Columbia, Vancouver, BC, V6T 1Z4, Canada
ewind@interchange.ubc.ca

ABSTRACT

Numerous studies across North America indicate that clearcuts and second-growth stands are unsuitable habitat for amphibians compared to old-growth forests. As landscapes become increasingly fragmented, amphibian populations in remaining old-growth patches are isolated. Insularization leads to decreased genetic fitness within populations and may lead to local extirpations or extinctions of species. Because amphibians have low vagility, high philopatry, and are susceptible to desiccation, they may be particularly vulnerable to habitat fragmentation. I present a general overview of impacts of forest practices and habitat fragmentation on amphibian populations, highlighting key management issues: forest patch size, importance of wet areas, and connectivity. I draw examples from studies that have specifically investigated effects of habitat fragmentation, clearcutting, and habitat juxtaposition to address the following questions: 1) What do we know about the impacts of forest practices and habitat fragmentation on amphibian populations?; and 2) What information is needed for resource managers to conserve amphibian populations in British Columbia?

Key words: amphibians, area effects, forestry, fragmentation, isolation, patch size.

AMPHIBIANS AT RISK?

Concern over amphibian population declines will be one of the biological legacies of the past decade. Initially scientists debated the issue, charging that populations fluctuate naturally, and that long-term data were needed before any conclusions could be drawn (e.g., Pechmann et al. 1991). More recently, however, many researchers have come to agree that the phenomenon is indeed real (Wake 1998). In some locations (e.g., western North America), amphibians are at far greater risk of extinction than birds or mammals (B.C. Conservation Data Centre 1998, Doyle 1998).

Amphibian species and populations that were once wide-ranging and common have been lost, or have declined, in what appear to be pristine habitats (e.g., *Bufo boreas* populations in the Colorado Rockies; Carey 1993, Drost and Fellers 1996). It has been proposed that the declines may be indicative of large-scale environmental decay, and research into specific causes has begun to shift towards those which may result in generalized stress (Wake 1998). Many populations may be exposed to factors (i.e., stressors) that under normal conditions do not cause them to decline (i.e., they appear "healthy"). When additional stressors are added, such as habitat alteration (e.g., forest harvesting), populations may be pushed past their tolerance limits. Further, the effects of stressors may be more pronounced in populations at the extremity of a species' range, making these populations more prone to extinction, but this has not been tested (Sjögren 1991*b*).

I suggest that all amphibian species in British Columbia may be at risk for the following reasons:

- The cause(s) of global declines in amphibian populations is not clear.
- The status of British Columbia amphibians is unknown (baseline data are lacking).
- Amphibians in the western United States are particularly vulnerable to extinction (Keister 1971), especially montane anurans (Drost and Fellers 1996).
- British Columbia amphibians may be particularly vulnerable to stress because many are at the northern limit of their range within the province (e.g., Pacific giant salamander [*Dicamptodon tenebrosus*] and Oregon spotted frog [*Rana pretiosa*]; the range of most of the Red-listed amphibian species in British Columbia is very limited; B.C. Conservation Data Centre 1998).
- Wide-spread habitat alteration is occurring throughout the province, with increased conversion of primary forest to second-growth stands, and greater isolation between remaining suitable habitat patches.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

EFFECTS OF FORESTRY ON AMPHIBIANS

Amphibians are ectothermic vertebrates capable of highly efficient energy conversion (Zug 1993). However, they must balance this energy efficiency with a physiological obligation to water. Many amphibian species lay their eggs in standing or running water, and all require moist environments for subcutaneous respiration (Zug 1993). Forests have moderate climatic conditions and structural attributes that provide amphibians with cover and moisture (e.g., downed wood). An example of this relationship can be found in the Pacific Northwest, where high amphibian species diversity and endemism is attributed to the combination of mountainous, forested areas and high rainfall (Keister 1971).

It has been demonstrated across North America that amphibian abundance is often lower in clearcuts and second-growth stands than in old-growth or mature forests due to changes in microclimate (e.g., Pough et al. 1987, Bury and Corn 1988, Petranka et al. 1994, Dupuis et al. 1995, Waldick 1997). However, it appears that not all species/populations are affected (e.g., Stelmock and Harestad 1979, Ward and Chapman 1995, Cole et al. 1997). It is difficult to know what the impacts of forest harvesting are on amphibians because most studies have been conducted within the past 15 years, which may be too soon to detect the long-term effects of habitat alteration (deMaynadier and Hunter 1995), and numerous logistical/design issues arise when attempting to compare studies (e.g., detectability of species within different habitat types). Reduced richness and abundance of amphibians in clearcuts and second-growth habitats at the stand level implies that these areas represent less suitable habitat types than old-growth forest, and that landscapes managed for forestry may increase isolation of populations, leading to genetic bottlenecks and/or local extirpation.

Landscape-level Effects of Forestry: Habitat Loss and Fragmentation

It is difficult to separate the effects of habitat loss from fragmentation (Fahrig 1997). A landscape becomes fragmented at different scales for different species depending upon whether the matrix environment poses a barrier to movement and dispersal. This results in either habitat loss or temporary/permanent habitat alteration. Amphibian populations may easily become isolated within a fragmented landscape because amphibians have low vagility, high philopatry, and often find the matrix environment inhospitable.

Amphibians may be particularly vulnerable to habitat fragmentation due to the spatial dynamics of their populations. Subpopulations of many species are philopatric to wet patches and upland forests. For this reason, some amphibian species live in metapopulations (e.g., many temperate aquatic-breeding species; Sjögren 1991*a*), likely in a variety of forms (Harrison 1991). For example, some species may live in relatively extinction-resistant populations, where inter-patch dispersal is fairly high. Others may live in a source-sink situation, where subpopulations remain viable only through the immigration of dispersing juveniles from source areas (e.g., Gill 1978). Metapopulations of the latter sort may be particularly vulnerable to local extirpation in managed landscapes, due to the fact that the high surface to volume ratio of juvenile amphibians makes them especially susceptible to desiccation (Bellis 1962), and hence, potentially impacted more by the removal of forest cover than adults (deMaynadier and Hunter 1999).

The dynamics of source-sink populations are poorly understood for amphibians, but have serious implications for the conservation of breeding sites and the maintenance of connectivity among populations. Source sites may not be the same for each species, and they may shift through time. Small wetlands are valuable breeding habitat for many amphibians (Gibbs 1993), and they may act as stepping stones for maintaining connectivity among populations. However, small wetlands and streams are not afforded protection within British Columbia. In some areas, it is not economically viable to protect all wet areas, and the selection of sites has been based on wetland size and the presence of fish. However, wetland size has been identified as a poor indicator of habitat quality and species richness (Gill 1978; D. Bradford, U.S. Environmental Protection Agency, and J. Jaeger and B. Riddle, University of Nevada, pers. comm.).

DO AMPHIBIANS RESPOND TO PATCH SIZE AND FRAGMENTATION?

Researchers have found that both amphibian abundance and species richness can be affected by forest area (Appendix 1). Hagar (1998) found that some amphibian species are area-sensitive in island archipelagos in Ontario, suggesting that the potential exists for some species to act as indicators of suitable reserve size and habitat fragmentation. In the most extensive long-term study addressing the effects of fragmentation on biodiversity, Tocher et al. (1997) found that amphibians did not respond to area effects as greatly as other taxonomic groups in tropical forest patches. In contrast, Rosenberg and Raphael (1986) found a greater stand-level response to fragmentation in amphibians than for birds or small mammals in California.

In other temperate studies, Dupuis and Bunnell (1999) found western redback salamanders (*Plethodon vehiculum*) responded to patch size in old-growth stands, but not in immature, second-growth stands. In Connecticut, Gibbs (1998*b*) found that some species were absent from a forest fragmentation gradient once forest canopy cover fell below 30–50%. Gibbs (1998*c*) also found that subpopulations of eastern redback salamanders (*Plethodon cinereus*) in a fragmented

landscape had greater genetic differentiation in locations where the forest had not been connected historically. Fragmentation may affect survivorship or reproductive success of amphibians (Wind 1996, Dupuis and Bunnell 1999).

Minimum Patch Size

Nested assemblages, where a few common species tend to be omnipresent and rare species tend to occur only at richer sites, occur in naturally and anthropogenically fragmented environments (Hecnar and M'Closkey 1997). In Ontario, amphibian assemblages have been found to be highly nested (Hecnar and M'Closkey 1997), which, combined with their area-sensitivity, indicates that large reserves should support more species than small ones (Hagar 1998). Rosenberg and Raphael (1986) and Wind (1996) identified amphibian species that negatively responded to forest patches 10 ha. In a 20-ha fragment in Michigan, Ball (1999) estimated that 2 salamander species had a thriving population, while 2 other species were potentially vulnerable or declining. The size of the patch that can sustainably support a population will be influenced largely by the degree of insularity in the landscape. Rosenberg and Raphael (1986) estimated that forest patches <20 ha in size lacked the full complement of vertebrate species, and they recommended that isolated stands with >50% insularity should be no less than 50 ha.

Influence of Habitat Features and Landscape Configuration

Many studies have found that amphibian responses to habitat fragmentation are influenced by the distribution of habitat features at the stand level (e.g., breeding sites; Appendix 1). The strong relationship between the distribution of amphibians and the patchy distribution of these habitat features may override area relations (Zimmerman and Bierregaard 1986). For example, Wind (1996) found that the existence of numerous wet areas in a 10-ha forest patch in Alberta appeared to mitigate area effects for wood frogs (*Rana sylvatica*) compared to drier 10-ha sites.

Just as the presence of wet areas will influence the distribution of aquatic-breeding species, the proximity to old-growth stands may affect terrestrial-breeding species within a multi-aged landscape. For example, Dupuis and Bunnell (1999) found that the density of western redback salamanders was lower only in managed stands that did not contain adjacent old-growth habitat, even though their patches were relatively large (i.e., >50 ha). In a study of 2 frog species in Equador, Marsh and Pearman (1997) found higher levels of abundance of both species in a large forest patch, but found that 1 species was affected by patch size while the other was affected by proximity to the large forest patch.

Edge Effects, Roads, and Movement Patterns

The impacts that edges have on amphibian populations are unclear. Just as for other taxonomic groups, amphibian response to edge habitats is variable. For example, Rosenberg and Raphael (1986) found greater species richness in landscapes with increased edge in California. In the Amazon, species bred in artificial pools irrespective of proximity to edge (Gascon 1993), and increased species richness in forest patches was due to an influx of species that responded positively to increased edge habitat (i.e., habitat generalists; Tocher et al. 1997). D. Major (USDA Forest Service, pers. comm.) found evidence of a bimodal response in terrestrial amphibians to silvicultural edges in Oregon. He attributed the response to blowdowns (i.e., increased availability of downed wood along edges), which potentially ameliorated effects of microclimate change through increased cover and moist microsites. In contrast to this, studies conducted in the northeastern United States have found reduced abundance of amphibians along edges compared to forest interior (Gibbs 1998*a*), with effects extending 25–35 m into the forest (deMaynadier and Hunter 1998). Microclimate changes along silvicultural edges may extend 240 m into the forest interior (Chen et al. 1995). Amphibians may respond differently to microclimate conditions along edges depending upon their diel patterns and biological needs (e.g., nocturnal versus diurnal; Schlaepfer 1998).

Moderately fragmented landscapes may facilitate the movement of individuals of some species, through the creation of roads and other linear corridors, and open habitats (Pither and Taylor 1998). Gibbs (1998*a*) found that the relative permeability of forest-road edges was less for amphibians than either forest-interior (controls) or forest–open land edges, and deMaynadier and Hunter (2000) found reduced abundance of salamanders at road sites versus forested control sites. Nadorozny (1997) radio-tracked ranids in Nova Scotia, and found that individuals would remain in roadside ditches, and follow roadways, when they were encountered. They also moved more through open habitats than forested ones. Contrary to the hypothesis that species with high dispersal ability should be less area-sensitive (see review by Hagar 1998), increased movement/dispersal ability for amphibians may increase sensitivity to fragmentation (Gibbs 1998*b*, Hagar 1998). Increased movement may increase risks of predation (direct effects), and/or result in greater energy expenditure, stress, etc. (indirect effects). Landscape-level conservation strategies must take into consideration these filters and conduits to amphibian movement (Gibbs 1998*a*), and their impacts on natal dispersal (deMaynadier and Hunter 2000).

CONCLUSIONS AND RECOMMENDATIONS

I conducted a review of the current literature to determine whether amphibians are affected by various measures of habitat fragmentation (Appendix 1). Because relatively few

studies have addressed this issue for amphibians, and the scales and study designs differ, it is difficult to draw conclusions at this point. In addition, Rosenberg and Raphael (1986) point out that habitat fragmentation is a relatively new phenomena in the Pacific Northwest, where a threshold level of habitat loss and isolation may not have occurred yet (Fahrig 1997). The long-term effects (i.e., an equilibrium) may not be detectable at this point. Given this, some amphibian species have responded to patch size and edges. Based on our limited knowledge at this point, I make the following recommendations to resource managers and researchers regarding habitat fragmentation and amphibian populations in British Columbia.

RESOURCE MANAGERS

- Plan for harvest units that are >20 ha (larger with increased insularity), and/or keep patches close together and close to old-growth stands.
- Maintain reserves of old-growth and mature stands, large enough to contain a heterogeneous mixture of terrestrial (upland) and aquatic habitats, deciduous and coniferous stands (Cole et al. 1997). Reserves require a minimum additional 35-m buffer to reduce edge effects and maintain the amount of interior forest habitat, and isolated patches of old growth <20 ha should not be included in the overall tally of habitat preserved as old growth (Rosenberg and Raphael 1986).
- Plan harvest units and roads so that the proportion of edge habitat is minimized, avoid placing roads near potential breeding sites (Vos and Chardon 1998), and do not apply chemical treatments (e.g., CaCl; see review by deMaynadier and Hunter 1995), especially during peak migration periods. Keep the number and width of roads to a minimum (deMaynadier and Hunter 2000).
- Identify key habitat attributes for amphibians, especially the location of wet patches. Plan harvest blocks in such a way as to leave wet patches in, or in close proximity to, old-growth and mature stands. Protect clusters of ponds (Sjögren 1991a), and ponds close to forest (Laan and Verboom 1990). Connect linear riparian areas to lentic and lotic systems, and upland forest (Dodd and Cade 1998).
- Harvested blocks should retain some large, live trees and snags to provide partial overstory shade to open clearcuts, and input of future sources of downed wood.

RESEARCHERS

- Design habitat suitability studies that include some measure of the "health" of amphibian populations, in addition to data collected on abundance and species richness. We need a clearer understanding of whether habitat is actually being lost (Fahrig 1997) or degraded to the point that amphibians in managed landscapes have reduced fitness (i.e., are no longer successfully breeding).
- Sample the distribution of habitat characteristics within the study area and beyond, especially when comparing habitat types/treatments that affect the distribution (e.g., downed wood and wet patches) and movement patterns (e.g., roads/streambeds; Nadorozny 1997, Dodd and Cade 1998, Gibbs 1998a) of amphibians.
- We need a clearer understanding of the dynamics of dispersal patterns, and barriers to movement, in order to reduce insularity among populations.

ACKNOWLEDGEMENTS

I thank P. deMaynadier, D. Major, and P. Taylor for their review comments. Also, thanks to those who provided information on current research. This is publication number M-14 for the Centre for Applied Conservation Biology.

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Appendix 1. Studies investigating measures of forest habitat fragmentation and amphibians.

Study Type	Location[a]	Objective(s)	Species	Design	Results/Conclusions	Source
Fragmentation	CT	Compared the population genetics of salamanders in fragmented and continuous forest	*Plethodon cinereus* (terrestrial-breeding salamander)	Forest patches within an urban/rural matrix	Genetic differentiation in fragmented population marginally greater than in continuous forest population; within subpopulations in fragmented landscape there was greater differentiation where forests were not connected historically.	Gibbs 1998*c*
Fragmentation	CT	Compared the distribution of woodland amphibians, with different life histories, along a spatially continuous gradient of forest fragmentation.	5 species (aquatic & terrestrial-breeding)	Forest fragmentation gradient, from heavily urban to rural environment	*P. cinereus* and *Pseudacris crucifer* found in suitable habitat along the gradient; *Rana sylvatica/Ambystoma maculatum*, and *Notophthalmus viridescens* were absent from portions of gradient with <30% and 50% forest canopy cover respectively. Traits of species with low tolerance to fragmentation were low density, population variability, and high mobility, coupled with restricted habitat needs.	Gibbs 1998*b*
Fragmentation	WA	Compared the richness and abundance of amphibians in stands of various sizes in relation to their surrounding landscape.	8 species (aquatic and terrestrial-breeding)	45 old-growth, mature, and young stands (ranging from 51-1689 ha) , centred within 2025-ha managed landscapes	No relationship was found between species richness and stand size. The abundance of *P. vehiculum* (and *Taricha granulosa*) was positively correlated with stand area. *P. vehiculum* may be packing into remaining stands. Stand-scale was more important than landscape variables. Elevation had a strong influence on distribution	Lehmkuhl et al. 1991
Area Effects	Brazil	Compared species richness of forest frogs in 1, 10, 100, and 500-ha reserves of primary forest	39 anuran species (aquatic and terrestrial-breeding)	Forest patches with clearcut / pasture / scrub matrix	Although a tight species-area relationship was found for forest frogs, the authors found greater explanatory power from the distribution of habitat features (i.e., breeding habitat) than area relations.	Zimmerman and Bierregaard 1986
Area Effects	Brazil	Compared richness, abundance, and breeding success within forest patches (100, 10, & 1 ha) and primary forest; pre and post-harvest data.	40 anuran species (aquatic and terrestrial-breeding)	Forest patches with clearcut / pasture / scrub matrix	The frog community appeared to less affected by habitat loss and fragmentation than other vertebrates; species richness was higher in patches, species response varied. Increased diversity was likely due to the influx of species which use the matrix habitat. Species presence was influenced by the distribution of breeding habitat and edges	Tocher et al. 1997

Appendix 1. Continued.

Study Type	Location[a]	Objective(s)	Species	Design	Results/Conclusions	Source
Area Effects	AB	Compared the relative abundance of frogs found in forest patches (10 & 100 ha) and continuous forest; pre and post-harvest.	*Rana sylvatica* (aquatic-breeding frog)	Forest patches within a matrix of young clearcuts	The relative abundance of wood frogs was reduced, and average size of frogs increased, in 10-ha patches 2 years after harvesting compared to control sites. Wetness of the site appeared to influence amphibian abundance in small fragments.	Wind 1996
Area Effects	BC	Compared the abundance and richness of amphibians in old-growth and managed second- growth forest patches, with or without adjacent old growth.	*Plethodon vehiculum* (terrestrial-breeding salamander)	Forest patches within a clearcut matrix	The relative abundance of redbacks was greater in large old-growth stands, but not in mature second-growth stands. Managed stands contained more salamanders, (perhaps with greater survivorship), when old growth was adjacent.	Dupuis and Bunnell 1999
Area Effects	MI	Estimated absolute abundance of amphibians in a 19.7-ha forest fragment with a history of logging activity as they migrated to breeding ponds.	3 *Ambystoma* spp. and *Notophthalmus viridescens* (aquatic-breeding salamanders)	Farmland on 3 sides, and a road on the 4th	*A. laterale* complex and *Notophthalmus viridescens* populations appear to be thriving. *A. maculatum* has few females in the population, and may be vulnerable to extirpation. Few *A. tigrinum* were caught, and likely does not represent a viable population.	Ball 1999
Area Effects / Edge	CA	Compared species richness and relative abundance between stands with varying degrees of isolation, edge, and area on different scales (i.e., plot, stand, 1000-ha block).	6 species (aquatic and terrestrial-breeding)	Forest stands, fully or partially isolated by clearcuts, second growth, or hardwoods	Species richness was positively correlated with edge and degree of fragmentation. Responses to amount of edge, proximity to clearcut, and patch size varied among species. *Dicamptodon ensatus* and *Bufo boreas* were species thought to be most sensitive to fragmentation, with no tolerance for stands < 10 ha.	Rosenberg and Raphael 1986

Appendix 1. Continued.

Study Type	Location[a]	Objective(s)	Species	Design	Results/Conclusions	Source
Area Effects / Edge	Ecuador	Compared the relative abundance of frogs within forest patches; at different distances from forest edge and within the matrix.	*Eleuthero-dactylus Chloronatus*, *E. Terpidotus* (terrestrial-breeding frogs)	Forest patches with pasture/low scrub matrix	The abundance of both species was greater in the large patch (200 ha) than small patches (0.25-5.3 ha). *E. chloronatus* was affected by patch size, whereas *E. terpidotus* was affected by distance to the large forest patch.	Marsh and Pearman 1997
Edge	Brazil	Compared the number of cohorts (i.e., signs of breeding) within plastic basins placed perpendicular to fragment edges (from inside forest, out to cleared area)	*Osteocephalus taurinus*, *Epipedobates femoralis* (aquatic-breeding frogs)	Edges of forest patches isolated by a matrix that was cleared for cattle grazing	For the two most abundant species, patterns of habitat use were independent of proximity to forest edge. *E. femoralis* used pools inside and outside forest at same frequency, however, *O. taurinus* did not breed outside primary forest.	Gascon 1993
Edge	CT	Compared an index of movement in relation to forest interior habitats, streambeds, edges and roads	6 species (aquatic and terrestrial-breeding)	Forest-road and forest-open land edges, and forest-interior controls	The capture rate of 3 species was influenced by forest borders; for all species, relative permeability of forest-road edges were much reduced compared to forest-interior and forest-open land edges.	Gibbs 1998*a*
Edge	ME	Compared the relative abundance of amphibians along silvicultural edges of different age classes	14 species (aquatic and terrestrial-breeding)	Edges of forest: 2- & 9- to 11-yr-old clearcuts, and forest: 5- & 25-yr-old pine plantations	Generally, salamanders were more sensitive than anurans, but habitat generalists and specialists were identified within each group. Depth of edge effects was estimated to be 25-35 m for management sensitive species (i.e., *P. cinereus*, *A. maculatum*,, *A. laterale*, and *R. sylvatica*).	deMaynadier and Hunter 1998
Edges of Forest Fragments	Costa Rica	Compared the relative abundance of frogs along forest-pasture edges	*Eleuthero-dactylus* frog species (terrestrial-breeding)	Cattle pasture-forest edges in primary forest fragments <25 ha	Nocturnal frog species were equally abundant along edges and interior forest, but one diurnal frog was less abundant along edges.	Schlaepfer 1998

[a] AB = Alberta, BC = British Columbia, CA = California, CT = Connecticut, ME = Maine, MI = Michigan, WA = Washington

The Yellowstone to Yukon Initiative: A New Conservation Paradigm to Protect the Heart of North America

Peter Aengst
Y2Y Initiative, 710 9th Street, Studio B
Canmore, AB T1W 2V7, Canada
paengst@banff.net

ABSTRACT

Recent research on animal movements and conservation biology theory indicates that our current protected areas are insufficient to sustain viable populations of large carnivores. A larger scale and more integrated approach is needed. Comprising over 100 conservation organizations as well as numerous scientists, the Yellowstone to Yukon Initiative ("Y2Y") is a bi-national effort to maintain and restore habitat integrity and connectivity along 3,200 km of the Rocky Mountains from the greater Yellowstone Ecosystem in the south to Yukon's Mackenzie Mountains in the north. Its goals are to ensure wildlife movements for all species between existing reserves as well as the creation of new protected areas to ensure population viability of all native species. Also proposed is a system of special management zones surrounding the core reserves in which carefully managed human uses may be permitted. Such an effort to promote ecological integrity can benefit human communities by diversifying economies, stimulating growth, and lessening boom-and-bust cycles of the past. Various mechanisms for creating this system are envisioned, including existing government land review processes, campaigns for new designated areas, private lands conservation, rail and highway crossings for wildlife, various cooperative programs, and development of a map-based conservation plan. As one of the few places in North America that still retains all of its native species, the Y2Y initiative presents an opportunity to proactively promote both healthy wildlife and human populations.

Key words: gray wolves, grizzly bear, national parks, protected areas, Rocky Mountains, Yellowstone to Yukon, Y2Y.

The Yellowstone to Yukon Conservation Initiative (Y2Y) is a joint Canada–United States effort of over 100 conservation organizations, wildlife scientists, planners, economists, and other individuals interested in protecting native wildlife, ecological processes, and wilderness in the Rocky Mountains of North America. This large-scale conservation effort seeks to establish a connected network of core reserve areas, transition zones, and wildlife movement corridors from the Greater Yellowstone Ecosystem in the south to the central Yukon in the north (Fig. 1). Existing national, state, and provincial parks and wilderness areas will anchor the system, while the creation of new protected and special management areas will provide the additional cores and corridors needed to complete it. This network will be based upon the principles of conservation biology, various focal species assessments, the knowledge of local and traditional residents, and the requirements for sustainable economies.

Footprints of human resource demands and development are today creating an archipelago of isolated natural areas in much of the Rockies. At risk is the unique large carnivore assemblage (e.g., grizzly bears, gray wolves, wolverines, lynx, mountain lions, fishers) that formerly existed throughout much of western North America and in the foothill prairies to the east. None of the national parks in the Rocky Mountains appears large enough by itself to protect viable populations of all large carnivores (Newmark 1987), with individuals typically entering several different land use designations in a year (Raine and Riddell 1991, Herrero 1995). Numerous studies have indicated that many of the free-ranging populations of large carnivores, "indicator" species of Rocky Mountain ecosystems, are now threatened (Mattson et al. 1992, Paquet and Hackman 1995). With the human population in the Y2Y region projected to double over the next 30–40 years (Willcox et al. 1998), the fate of these species will be decided by the land management decisions and conservation actions taken today.

The comprehensive network that Y2Y envisions is thus a response to these alarming trends. It is based upon a large-scale, long-term strategy recommended by many conservation

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

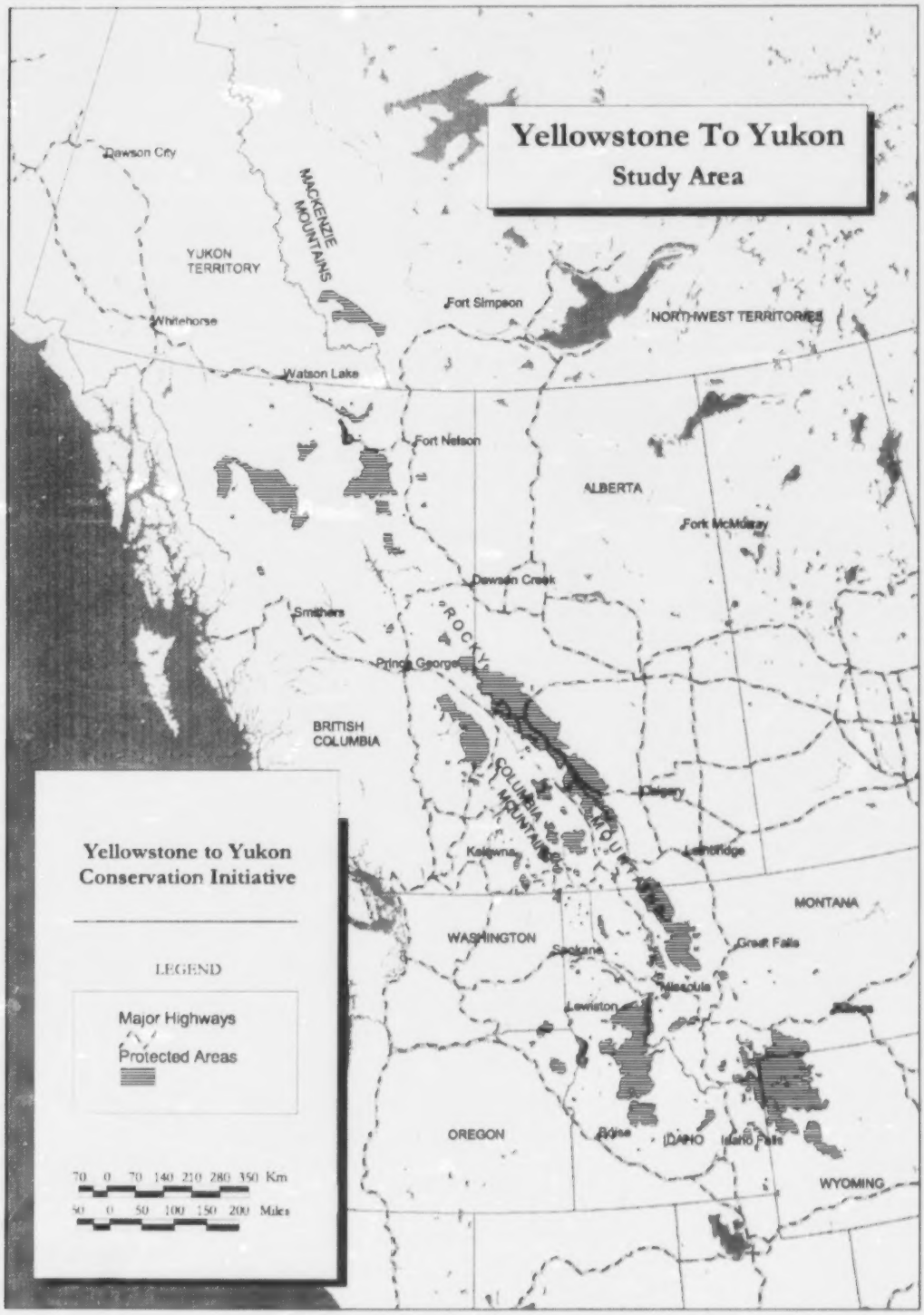

Figure. 1. Yellowstone to Yukon Initiative study area.

biologists (Shaffer 1992, Noss and Cooperrider 1994, Paquet and Hackman 1995) and which is currently being used in conservation proposals for the state of Florida, the "sky island" ecosystem of the southwestern United States and northern Mexico, the coastal rainforests of British Columbia, and in parts of Central and South America. Under this strategy, the core protected areas would provide large, secure refuges for all native species and accommodate nonindustrial human use consistent with the refuge goal. The corridors would ensure the landscape is permeable so there are no barriers to genetic exchange among subpopulations of any species. The transition zones would allow for human use of increasing intensity as one moves away from the core areas.

The Y2Y region also contains a variety of communities that are home to many people. The Initiative thus supports community development that is consistent with the extraordinary conservation values of the area. It envisions healthy communities based on sustainable economies as opposed to being wholly dependent on extractive industries. Indeed, a fundamental principle of Y2Y is that its effort to promote ecological integrity can benefit human communities by diversifying economies, stimulating growth, and lessening boom-and-bust cycles of the past.

METHODS

The thinking behind Y2Y is based upon several key lessons from wildlife research conducted over the past 2 decades. By mapping carnivore movements and home ranges, researchers have been able to delineate wildlife usage of the landscape and the importance of a contiguous wildland matrix, interconnected by habitat corridors linking core protected areas. The large-scale result of this carnivore mapping exercise delineates an area from the Greater Yellowstone Ecosystem to the Peace River country in the northern Rockies of British Columbia (Clark et al. 1996). The inclusion of the Yukon recognizes the importance of large undisturbed areas as source areas for wildlife in the greater Rocky Mountain region—as well as the fact that little of the key habitat in the Yukon is currently protected.

Furthermore, this mapping research has demonstrated that many of the keystone and indicator wildlife species of the Rockies travel much greater distances and thus require much more protected habitat than was previously realized. One study (Hummel and Pettigrew 1991) concluded that maintaining a viable population of as few as 390 grizzlies in the Rockies may require an area of 81,430 km^2—over 9 times the size of Yellowstone National Park. Also evident is the heightened importance of fostering connectivity between our protected areas. Research into wolf movements has shown that dispersal distance of individuals looking for a mate or territory can be hundreds of kilometres. One wolf was tracked from the Flathead Valley in Montana to Mile 0 on the Alaska Highway—a straight-line distance of 840 km (Boyd et al. 1995).

Another key lesson stemming from this wildlife research is that many predators on the top of the food chain act as keystone species, playing an important role in maintaining the integrity of the entire system (Terbough 1988, Mitchell 1998). Evidence collected by ecologists indicates that protection of habitat for these top-level predators can provide an umbrella for other species' needs. One study (Paquet et al. 1996) in the central Rockies of Canada found that by protecting habitat for the gray wolf, the complete habitat needs of 334 of the 381 native vertebrate species were taken care of and habitat for another 32 vertebrate species were partially addressed. This leaves the habitat needs of just 15 vertebrate species unaccounted for by protecting sufficient wolf habitat.

Finally, research into island biogeography has demonstrated that isolated populations tend to eventually die out because of greater fluctuations in population numbers, problems of genetic inbreeding, and higher mortality rates from disease and natural disturbances (Noss et al. 1996). This research further indicates that reserve size, inter-area distance, shape, number, buffering, and connectivity are critical factors in determining long-term population viability (Soule and Simberloff 1986). The current status of the grizzly bear in the United States provides clear evidence of the isolation problem. Currently less than 1,000 bears, or 1% of their original numbers, survive in 6 shrinking islands of habitat—all located within the Y2Y region (Shaffer 1992). Despite significant efforts to protect these bears from illegal killing and food conditioning, many biologists believe that these numbers have not increased appreciably since the populations were listed as threatened in 1975 (Wilkinson 1998).

Rationale for Y2Y

Clearly, a fundamental disconnect exists between many of the findings of wildlife researchers and our current protected areas system. While the conservation accomplishments of our ancestors in setting aside national parks, creating game refuges, and regulating sport hunting were critical and remain impressive in many ways, the symptoms of a growing biodiversity crisis in the Rocky Mountains remain (Table 1). A variety of factors have contributed to this current state of our protected areas system in the Rockies not fully working from an ecological standpoint (Table 2).

A primary problem has been the limited habitat and ecological features represented in most protected areas. Land protection in much of the Rocky Mountains is skewed heavily toward higher elevation areas ("rock and ice") that do not support many species or function to link populations. The low elevation valleys and riparian areas that contain the highest biodiversity and upon which many wildlife species rely for winter range and migration are instead being rapidly developed and fragmented over much of the region. An

Table 1. Symptoms of growing biodiversity crisis in Rocky Mountains.

- Loss of biological connectivity across the landscape.
- Increased isolation of habitats, wildlife populations, and ecosystems.
- A continual decline in wildlife numbers, especially large carnivores.
- Impairment and simplification of habitat stemming from suppression of ecological forces (wildfire, floods, etc.) and introduction of non-native species.
- More conversion of natural landscapes to humanscapes.

Table 2. Key problems with current protected areas system.

- Historically areas protected primarily for aesthetic and recreational reasons. Thus, much of the resulting protected habitat is lower value "rock and ice."
- Keystone species and natural disturbance regimes still suppressed—even in protected areas.
- Mortality and fragmentation associated with transportation corridors are not effectively addressed.
- Political boundaries and management objectives make little ecological sense.

enduring features gap analysis of the Y2Y region concluded, "Even though this region of North America is known for its spectacular national parks, the physical variability in the region as a whole was found to be poorly represented. High elevations were found to be strongly emphasized in the region with many low elevation landforms totally missing" (Strittholt and Frost 1997). This finding has direct consequences for wildlife. A recent study (Gibeau in press) of grizzly bear habitat use in the upper Bow River watershed found that the most secure grizzly bear habitat is located in the land jurisdictions with the least amount of protection and the higher zones of human use.

Another important factor is the status of keystone species such as wolves and grizzlies. Historical persecution has suppressed their numbers in many areas. Yet, though these species now receive some level of protection, they are still often not allowed to play out their critical ecological role. Even our protected areas system has problems. In Banff National Park, development around the town of Banff has prevented wolves from preying on resident populations of elk and mule deer. The result has been unnaturally high ungulate numbers resulting in increased herbivory on aspen and other detrimental ecological effects (Page et al. 1996).

Roads and railways inside and between our protected areas also impact on wildlife. Human transportation corridors act as wildlife mortality zones that block migration and genetic and demographic exchange between populations. One study found that 523 large animals were killed on Canadian Pacific Railway's main line from 1981 to 1998 in the Bow Valley alone (Gibeau and Heuer 1996). Across the Y2Y region, 8 permanent highways and rail lines run east–west bisecting habitat and wildlife travel routes (Willcox et al. 1998). Major transportation corridors such as the British Columbia/Alberta Highway 3 and the Trans-Canada Highway may be critical population-fragmenting elements for grizzly bears inhabiting the Rocky Mountains, which primarily run north to south. For example, despite construction of wildlife underpasses and overpasses on the Trans-Canada Highway in Banff National Park, the highway remains a total barrier to crossing by adult female grizzly bears (Gibeau and Herrero 1997).

A final factor worth noting is the challenge of political boundaries and jurisdictions not operating at the same scale as ecological processes necessary to support wildlife populations. Whether at the international, provincial/state, or local level, the laws, regulations, and agency practices work at odds with regional conservation. Consider the international border running through the Flathead Valley of northwest Montana and southeast British Columbia. Represented by a "cutline" a few metres wide, a grizzly bear can wander across it in seconds. The same habitat and grizzly population exists on both sides. Yet, to the south of the line, the bear is a protected species under the U.S. Endangered Species Act. An individual caught shooting a bear there would face a fine of up to $50,000 and 10 years in prison. But once the bear crosses north of this line, it can be shot as a legally hunted animal.

Y2Y's Approach

Y2Y is different than traditional approaches to conservation in many ways. First, Y2Y's ecosystem-based approach recognizes people and their cultural traditions and views humans as a fundamental component of the natural community. Y2Y is also different in that it is science based and focused on protecting connections—both the habitat connections many species need to continue to thrive and the connections to the natural world that supports much of the economy in this region, our quality of life, and our cultural identity (see Rasker and Alexander 1997). Finally, Y2Y is different in that it is a departure from the problematic and common mode of reacting in a piecemeal way to various site-specific threats and trying to work only through government-defined processes that operate in response to political jurisdictions and realities. Instead, Y2Y is proactive and holistic, and proposes to set up a citizen-led effort defined upon ecological boundaries and realities.

The on-the-ground specifics of the Y2Y proposal are still evolving. To plan, gather, and organize on a more workable scale that still makes ecological sense, the Y2Y region has been subdivided into 7 smaller areas (Table 3). Over the next 2 years, each region will undertake a conservation planning process that will result in a series of regional map-based

Table 3. Y2Y subregional conservation planning areas.

1. Greater Yellowstone Ecosystem: Yellowstone and Grand Teton national parks; 7 surrounding national forests; and 3 national wildlife refuges.
2. Greater Salmon–Selway/Hells Canyon: Central Idaho Wilderness Areas and Hells Canyon region in eastern Oregon and Idaho.
3. Crown of the Continent Ecosystem: Waterton Lakes–Glacier Peace Park; the Bob Marshall Wilderness complex; Rocky Mountain front of Montana; Akamina–Kishenina Provincial Park; Flathead Valley; and Castle–Crown area.
4. Central Rockies/Yellowhead Ecosystem: Banff–Yoho–Kootenay–Jasper Park complex; Kananaskis Country and White Goat/Siffleur, Willmore, and Ghost River wildernesses in Alberta; Mount Assiniboine, Height of the Rockies, Elk Lakes, Mount Robson, and Hamber provincial parks in B.C.
5. Interior Rainforest: Cabinet/Yaak/Selkirk ecosystem of Idaho; Greater Purcells (Purcell, Valhalla, and Kokanee Glacier parks in B.C.); and Columbia Mountains Ecosystem (Glacier, Revelstoke, Bugaboo, and Monashee parks)
6. Northern B.C.: Fraser Basin (northern Rocky Mountain Trench); Kwadacha and Spatsizi parks; Muskwa–Kechika protected area; and Cassiar Range.
7. Yukon/Northern Rockies: Liard Plain and Hyland area on B.C.–Yukon border; Mackenzie Mountains in western NWT; Selwyn Mountains; and Wind, Snake, and Bonnet Plume watersheds in the Yukon.

proposals that identify the proposed core reserves, habitat linkages, and transition zones. Using a geographic information system (GIS), the process will incorporate the principles of conservation biology, the best available wildlife habitat and population data, and the knowledge of those locals who know the land best. Following scientific review, the regional plans will be compiled to form a Y2Y-wide proposal that will be continually updated and revised, as new and better information becomes available. An inclusive and transparent process will be used in all stages.

To implement this map-based proposal will require various mechanisms including the potential use of existing government planning processes. Connecting and expanding the current system of federal, state, provincial, and privately protected lands will involve a mixture of economic, political, administrative, and legal decisions and incentives. More than new land designations will be needed. Transportation wildlife mitigation structures, restoration efforts, municipal growth management plans, land acquisition and exchanges, conservation easements, and cooperative management agreements are just some of the other mechanisms that could be part of Y2Y's regional conservation plans.

To minimize the need for new laws and planning processes, Y2Y will incorporate the results of existing land use processes in its planning, granting that the results reflect the needs of wildlife and support ecosystem integrity. The Fort Nelson and Fort St. John (British Columbia) Land Resource Management Plans (LRMP) which were completed in late 1997 demonstrate how a planning process can dovetail with the spirit and intent of the Y2Y initiative. In that case, Y2Y leaders were able to work through the LRMP stakeholder roundtables to achieve a connected system of new protected areas and special management areas, which serve as transition zones, in the Muskwa-Kechika area of north-central British Columbia (Hume 1998).

CONCLUSION

Clearly, a comprehensive conservation vision for the Rocky Mountains of North America is needed—to protect the remaining critical habitat and to ensure connectivity in this mosaic of mixed patterns of land ownership and land use management objectives. Y2Y's large, connected network of core protected areas, movement corridors, and transition zones would restore species' natural pathways and reconnect wildlife populations across the backbone of the continent. It would also promote natural ecological processes and foster the transition to more sustainable economies in the region.

The Y2Y Initiative is among the largest scale conservation efforts ever undertaken in North America. It is obviously an ambitious undertaking in both geographic and human terms. Yet, the Y2Y landscape—the heart of North America—is truly a special place. A symbol of the wild across the world, this region is the last refuge for many of the great carnivores, for many ungulates, for sensitive fish species, and for vast untrammeled watersheds. It is certainly a place worthy of our best efforts at protection and restoration. If we cannot save wildlife and wilderness in the Rocky Mountains of Canada and the United States, 2 of the most prosperous countries on earth, there is little hope for either in the world.

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St. John River Valley Hardwood Forest of Western New Brunswick: Developing Conservation Initiatives for an Endangered Forest Assemblage on Private Lands

A. MacDougall
3156 West 42nd Avenue, Vancouver, BC, V6N 3H2, Canada
asmacdougall@sprint.ca

ABSTRACT

Consideration of socioeconomic factors is an important component in the implementation of conservation initiatives on heavily managed landscapes. Ownership, prevailing economic climate, regional population trends, and land use history all influence the distribution and abundance of rare elements, as well as the procedures leading to protection. Saint John River Valley Hardwood Forest (SJRHF) is a relic disjunct forest assemblage of western New Brunswick that contains 43 floral taxa listed as rare in northeastern New England and Atlantic Canada. Two centuries of land clearance have reduced this once-continuous assemblage to a series of small (mean = 11 ha) and highly isolated fragments covering < 1% of its original land base. Recent inventory work has broadened knowledge of the SJRHF natural history and range occurrence in New Brunswick. However, ownership and related management issues have emerged as the most critical conservation factors. 99% of identified high priority SJRHF stands were privately owned, often by multiple owners. Most owners had lived on or near the property for some time and practiced limited selection harvesting. Low intensity harvest inadvertently protected these sites; owners had little or no knowledge of biodiversity values. Recent economic pressures, including increased hardwood demand and higher prices, improved harvest technology, government initiatives to increase agricultural production, and changes to wood supply access for private harvesting contractors have led to increased cutting pressure on SJRHF stands. While contacted owners expressed interest in maintaining rare flora on their property, most were wary of acquisition-based protection initiatives. As a result, SJRHF conservation measures will require a diversified owner-specific approach that includes increased information dissemination and the development of modified harvesting strategies. Initial attempts at the latter have proven difficult due to lack of knowledge of rare flora disturbance thresholds and resistance by local contractors to embrace alternative harvest methods.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Ecosystem Plan for Pelee Island

Mark R. Forbes
Department of Biology, Carleton University
Ottawa, ON, K1S 5B6, Canada

Gina Schalk
Department of Biology, Carleton University
Ottawa, ON, K1S 5B6, Canada

Sharleen M. Hawco
Department of Biology, Carleton University
Ottawa, ON, K1S 5B6, Canada

Kent A. Prior
Endangered Species Division
Canadian Wildlife Service
Hull, PQ, K1A 0H3, Canada

ABSTRACT

Populations of no fewer than 12 plant and 8 vertebrate species of national significance exist on Pelee Island, a 4,000 ha land base in Lake Erie, Ontario. Strong negative interactions between many of these species and various human activities such as agriculture, quarrying, recreation, and residential development suggest that recovery strategies must address both ecological and socioeconomic concerns simultaneously across the island landscape if they are to be effective. Pelee Island is now a mosaic produced by different land use practices, many of which are mutually incompatible (e.g., vineyards, savannah restoration). The management of this mosaic is also disassociated, being the responsibilities of different land owners and agencies with specific interests. However, many components of this landscape depend on one another. For example, blue racers (*Coluber constrictor*) have specific habitat requirements that oblige them to move among, and interact with, different landscape components (i.e., hibernation in limestone plains, foraging in agricultural fencerows, nesting in shoreline logs). Elements in a land use parcel are very rarely restricted to that parcel, but impact on adjacent segments. Managing individual landscape segments in isolation tends to ignore these interactions and has led to serious problems. Our contention is that ecological and socioeconomic factors cannot be isolated from one another if we are to ensure durable recovery of endangered species and the Pelee Island ecosystem. As such we are developing a coordinated Ecosystem Plan for Pelee Island (EPPI), an integrated plan that arises from the recognition that different segments and land uses are closely interlinked. During our preliminary field season in 1998 we set out to accomplish 6 key objectives: (1) establish EPPI database and collate existing data; (2) identify focal species and processes to serve as indicators of ecological integrity; (3) initiate data collection on focal species, ecological processes, and socioeconomic elements; (4) conduct spatial analyses to identify and prioritize options for ecosystem recovery; (5) develop ecosystem monitoring plan with standardized protocols for future data collection; and (6) identify key decision rules and develop a framework for EPPI. Our presentation reports on progress to date.

Key words: ecosystem recovery, extinctions, focal species, Pelee Island, population monitoring.

Two key motivations inspired this study. The first was the need to develop a national framework for coarse-scale recovery efforts to complement traditional single species focused recovery. In this sense, we hope that Pelee Island might represent a model or case study for ecosystem-level recovery in Canada. Such an approach aims to consider the collective requirements of co-occurring species at risk, interactions with human activities, and the need to enhance or restore ecological processes and functions. The second reason for this study was that Pelee Island in Lake Erie is a well-delineated hot spot of endangered species in Canada—in itself, deserving conservation monitoring and recovery action. Biologically, Pelee Island is perhaps the most rich parcel of land in Ontario. Work coordinated by Ontario's Natural Heritage Information Centre (NHIC) has identified >100 rare plant species on the island. Five species have been listed nationally by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC).

In addition to being a biological hot spot, Pelee Island is a microcosm of the typically competing interests of human land use practices and ecological conservation. Less than 20% of Pelee Island's 4,400 ha can be described as natural habitat while agriculture, limestone quarrying, hunting, along with resort and cottage development, dominate most of the landscape.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

PELEE ISLAND: HISTORICAL, BIOLOGICAL, AND SOCIAL CONTEXT

Pelee Island is the largest of a group of islands in the western basin of Lake Erie, and is the southern-most inhabited point in Canada. It is roughly rectangular in shape, 14 km long by 5 km wide. Historically, it consisted of 4 separate islands connected by sandbars containing a large interior marsh. In the 1880s, a series of dikes and canals was constructed to drain the marsh and expose land for farming, thus creating the single, consolidated island we have today.

The permanent human population peaked at about 1,000 in 1913. Current agricultural land-use includes a thriving wine industry, limestone quarrying is expanding, and growth of recreation and tourism show no signs of levelling off. At present, 3 large-capacity car ferries service the island which inflates the summer population to around 3,000 people. Land use planning and management is currently uncoordinated and the many different stakeholders have often non-complementary interests.

Pelee Island has a unique biological diversity. Ontario's NHIC has designated >100 plant species on the island as rare locally, provincially, or nationally and COSEWIC lists 12 of them as at risk of extinction including eastern prickly pear cactus, heart-leaved plantain, blue ash, American water willow, red mulberry, and prairie rose. Vertebrates currently listed by COSEWIC that reside on Pelee Island include smallmouth salamander, blue racer, Lake Erie water snake, eastern fox snake, spiny softshell turtle, Acadian flycatcher, and yellow-breasted chat. Pelee Island is known to have lost at least 12 vertebrate species through extinction of local populations including the least bittern and piping plover, a rich suite of snakes, and several frogs.

MAIN OBJECTIVES

We had several interrelated objectives for this first field season on the island. First, we wanted to establish the framework for a long-term monitoring program that would enable us to track changes in plant populations throughout the system. We feel that empirical quantification of change is critical if we are to evaluate responses of populations to specific management actions or disturbances. Therefore, we were mainly concerned with spatial distributions and abundances of rare plant species on Pelee Island in relation to several factors, including their representation at previously surveyed sites, their abundances, and their representation relative to socioeconomic or human-intrusion factors. In general, we also wished to determine whether the documented vertebrate extinctions are being mirrored by extinctions in plant populations. How imperilled is this ecosystem? The assumption is that things are very bad but to date no data support the contention. To investigate the status of plant populations we chose to compare historic and current distributions of selected species across the island. We inventoried 6 focal rare species. The 1998 field season allowed us to assess changes in distributions of the focal species, establish baseline measures for species abundances, and quantify the relationship between focal species and human disturbance factors. We also assigned conservation priority to specific species or sites on the island.

Choice of Rare Species

Of the >100 plant species identified as rare, we chose 6 focal rare species for closer examination in the 1998 field season. Our criteria for choosing species began with selecting species that have received NHIC priority ranking of 1–3. We then selected species that, although rare, are adequately represented on Pelee Island because our focus was on potential for decline in populations, rather than attempting to examine causative (historical) agents for restricted distributions of the rarest species. Information on occurrence and distribution was obtained from the extensive NHIC database. Finally, we narrowed our choice of rare species based on logistical considerations: we chose plant species that were easily identified by nonexperts in the July/August field season.

Focal Species

Our final choices of rare species consisted of 3 herbaceous species (Short's aster, *Aster shortii*; woodland lettuce, *Lactuca floridana*; and gray-headed coneflower, *Ratibida pinnata*); 1 shrub (prairie rose, *Rosa setigera)*, 1 tree (red mulberry, *Morus rubra)*, and 1 wetland species (leafy blue-flag iris, *Iris brevicaulis)*.

Associated with our sampling of these species we addressed the following questions:

- Do focal rare species show current distributions that are incongruent with historical distributions (i.e., local extinctions or expansion events)?
- Do focal species show either negative or positive relations with socioeconomic factors on Pelee Island?
- Based on our censuses, which of the focal rare species need future monitoring and/or recovery action most?
- Are any observed population declines more likely explained by species-specific problems or by general problems on Pelee Island?

Data Collection and Analysis

To address our questions, we first selected about 10 sites per rare species. Sampling locations were concentrated within 7 "natural areas" including Lighthouse Point & Fishery Woods, Brown's Road Savannah, Middle Point Woods, Municipal Campground and Stone Road Alvar, Fish Point, Dump Cedar Savannah and Red Cedar Savannah, and Sheridan Point.

Four of these areas were extensively inventoried by others previously, and their conservation significance can be seen

easily by the numbers of nationally rare plants known from each area: Lighthouse Point (29 species), Middle Point Woods (30 species), Stone Road Alvar (49 species), and Fish Point (52 species). We added 3 other natural areas to our inventory, but also relied heavily on the previous inventories for the baseline against which changes in representation of species could be assessed.

Our sample sites (or EPPI sites) were chosen from within these 7 natural areas because other tracts of land, which showed extensive agriculture or other land use practices, were unlikely to support viable populations of the focal rare species. Again, our main objective was to see how well populations of species were doing in more natural areas and whether any populations showed historical declines or negative or positive relations with human intrusion. For these reasons, we tried to distribute sites among natural areas for each species, although this was not always possible (e.g., red mulberry occurred only at Fish Point in 1998).

EPPI Plot Sampling

At each EPPI site, we marked out a 0.25-ha plot. For each plot, one 25-m transect was marked in a north–south orientation and another was marked in an east–west orientation, to make a "+". Universal Transverse Mercator (UTM) coordinates were recorded from centre of each plot. We recorded number of individual plants of each focal rare species within sight along the length of each transect, and noted the quadrant of the plot in which the species occurred. Rare species were scored according to the intensity (number of plants in each plot) and extent (number of quadrants in which plants occurred) within each plot.

In each plot, we also recorded the intensity of human intrusions based on the evidence of logging, farming, livestock grazing, mowing/clearing, planting, tracks and trails, dumping of rubbish, earth displacement for development, recreational use, and hunting. We then scored each plot for intensity and extent of human intrusion.

RESULTS

Our first question was addressed simply by comparing presence of focal rare species at these sites with what had been reported historically. Of course, such a comparison requires we revisit exact locations visited historically by NHIC personnel. We chose only those sites where a high degree of accuracy in assigning a location to the site was indicated in NHIC reports (from 1982 to 1997). Thus, the absence of a species from a site where it was previously located would represent a local extinction.

Fifty-seven EPPI sites were selected for detailed analysis based on the occurrence of one or more of the rare plant species in previous surveys. Of the 76 EPPI site-species combinations (some sites had more than 1 rare species), rare species were relocated at sites about 82% of the time. However, repeatability was variable across species: it was highest for red mulberry (100%) and Short's aster (90%). We found red mulberry in all the places that we expected it. By contrast, leafy blue-flag iris had the lowest reoccurrence at 37.5%, meaning that it disappeared from >60% of the locations in which it previously occurred. In fact, 2 populations of iris went extinct during this study due to cottage development. Woodland lettuce, grey-headed coneflower, and prairie rose also declined. Interestingly, Short's aster was found at several locations sites where it had not previously been recorded, indicating high dispersal capabilities of this species.

As with persistence, relations of focal rare species to socioeconomic factors were also variable. Over all EPPI sites, tracks and trails and hunting were the most common activities. To assess potential influence of socioeconomic activities on rare species' abundances, we conducted a series of Spearman rank correlations. Based on these analyses, none of the rare species showed significant correlations with the composite socioeconomic scores. However, our composite measure may obscure important information about individual types of socioeconomic activity

To explore this relationship further we examined the distribution of socioeconomic activity at sites with loss of at least one focal rare species, compared to sites with no loss of focal species. Sites that showed no losses were concentrated at the lowest range of socioeconomic scores (scores 1–5). At slightly higher socioeconomic activity (scores 6–10), sites with losses were more numerous than sites with no losses (10 vs. 4 sites, respectively). These results suggest that focal rare species tend to persist at sites with the lowest socioeconomic activity compared with sites with slightly higher activity.

Taken together, these results on single species suggest some priorities for further monitoring and conservation. First, of the 6 rare species we studied, only 2 were represented at ≥90% of sites censused in previous surveys. These 2 species were the red mulberry and Short's aster. At the start of the study, we assumed that presence–absence data for red mulberry would not be informative, although some have suggested its abundance on the island has declined dramatically based on earlier surveys. We included red mulberry to provide a baseline for further studies on its abundance. Other metrics such as recruitment of new saplings, ratios of hybrids to parental plants, or measures of stress on older trees will probably be more informative. Thus, only Short's aster showed considerable permanence. It also was sampled from 12 new sites and is an extremely vagile species. The other 4 species showed less permanence. For woodland lettuce, there is some compensation: our data suggested losses at 4 sites but also gains at 2 sites. However, leafy blue-flag iris had substantial losses. We are certain that a decline in representation on the island is still occurring for this species,

especially considering that 2 sites no longer exist where this species has been known to occur.

An appropriate null model is that changes in species distribution have random direction and that overall, the number of sightings is more or less constant from year to year. Thus, most sites are expected to show species formerly censused there and the sites showing change should be more or less equally represented by species losses and species additions. While this might be the case for woodland lettuce, it is not the case for the remaining 3 species, which showed only losses. Extinction events are expected for small and isolated populations. Such patterns of losses, if repeated across other plant species at risk on the island, would clearly indicate decline in ecosystem integrity measured using plants. Such declines may be following the earlier declines and local extirpations of vertebrates on the island.

Few other relations were found between socioeconomic factors measured at EPPI sites and scores of rare species abundances. In this study, socioeconomic factors were measured as anthropogenic intrusions into natural areas where plants were censused. By definition, these intrusions were small (otherwise the habitats would not be described as natural). Such intrusions are similar to environmental or demographic stochasticity (i.e., they may be important only occasionally). Small populations can have high baseline probabilities of extinction due to likelihood of recolonization or rescue by healthy populations elsewhere. After extinction has occurred, the anthropogenic intrusions may hinder the realization of healthy populations nearby, especially if they act as sinks of inhospitable habitat.

In summary, our approach thus far has been to classify rare species based on 4 criteria: (1) their representation at sites, (2) their relative abundance (score) across sites, (3) their historical changes in distributions, and (4) their relation to socioeconomic factors. Based on these criteria, only Short's aster is not in immediate threat of further reduction in its distribution or abundance on Pelee Island (although listed as vulnerable by COSEWIC based on its restricted distribution in Canada). While red mulberry has not decreased its presence at sites on the island, it was formerly regarded as abundant. Others have suggested that hybridization and introgression with white mulberry may threaten red mulberry populations on Pelee Island. The other focal rare species, including leafy blue-flag iris, gray-headed coneflower, prairie rose, and woodland lettuce, may show reductions for similar general reasons (habitat loss and fragmentation). For example, the degree to which a natural habitat is surrounded by uninhabitable farmland or terrestrial habitats may play a role in the size and viability of rare plant populations within the natural area. Rates of local extinction should be slowed if other "source" populations exist near to the island, or if the species of interest is vagile.

Whether decline in representation of rare plants on Pelee Island is a more general phenomenon and whether it is related to general problems of habitat loss and fragmentation, a suite of species-specific causes or a combination of both general and specific factors will have to await monitoring of other species at risk. From this study, however, some species are in decline and EPPI sites housing those species should be protected (particularly for leafy blue-flag iris).

RECOMMENDATIONS AND STEPS TOWARD CONSERVATION

Recommendation 1

Current EPPI sites where focal rare species were not recorded, but were expected, should be revisited for future monitoring to determine whether recolonization occurs.

Recommendation 2

Current EPPI sites from each natural area should be chosen for further monitoring. Rather than choose sites with high species diversity, sites with low species diversity and low abundance (those most susceptible to local extinction) could be prioritized, because general factors causing declines, if present, should be most apparent at such sites.

Recommendation 3

New EPPI sites from each natural area should be chosen for monitoring of other focal rare species to determine the generality of declines. This objective was beyond the scope of our 1998 field season because we had only 1 dedicated technician in the field.

Surveys based on other species are critical to assess whether declines, similar to those for vertebrates, are general for plant species at risk. Such information would further highlight the need for immediate recovery action. Species that are representative of aquatic and various terrestrial habitats need to be chosen. These species also should be readily identifiable by nonexperts. Correlations between these other species and species chosen in this study would help assess the suitability of some species as indicators. Finally, species chosen for future study should include both insect- and wind-pollinated species. Declines in pollinator diversity or abundance have been cited as potentially contributing to declines of certain plant species. It is currently unclear whether pollinator declines expected elsewhere in Ontario (due to *Varroa* sp. parasitism) also affect Pelee Island.

Recommendation 4

Recovery action should be undertaken for leafy blue-flag iris. This species is ranked high by the NHIC, is extremely rare nationally, and is potentially showing great declines. Two of the surveyed sites have already been destroyed by cottage development.

Recommendation 5

Reseeding programs should be considered for leafy blue-flag iris and other focal rare species to determine whether recolonization is limiting for species showing declines.

Current EPPI sites where leafy blue-flag iris is present should be protected by limiting visitor access. Any prospective cottagers should be made aware of effects of development and other practices (e.g., wharf building) on iris habitat. Any reseeding initiatives should first assess whether healthy populations of the target species show strong seed sets elsewhere on the island and then harvest some seeds (not plants) for seeding at sites on the island where that species was found previously, but no longer exists. This work is critical for determining whether the habitat can support the species of interest, and whether recolonization has been limiting. Recovery actions aimed at reseeding plants and establishing several distinct populations of each target species may then be undertaken on the island at relatively low cost. Plants should be reseeded near cottages, where habitat is known to have been suitable. If reseeding programs fail, then some habitat may have to be created (e.g., ponds with margins of emergent vegetation as habitat for leafy blue-flag iris).

Recommendation 6

Every attempt should be made to protect the remaining natural areas on Pelee Island. Although few species showed negative relations with socioeconomic factors, we believe that, at the larger scale of the island, habitat loss and fragmentation may be causing general declines among plant species at risk. This expectation is consistent with data collected in this study, and other empirical and theoretical work. Further, sites where species have been lost appear to have higher socioeconomic scores than sites where species were resighted in 1998. Taken together, these data urge caution about further monitoring and slowing or halting development of natural areas on the island. Further attempts should be made to develop eco-tourism alternatives for Pelee Island. Any development initiatives through the town council should be made public, so that ideas for alternatives to development, or conservation actions given development, can be expressed by individuals or organizations.

In summary, these recommendations are aimed at further monitoring and recovery of plant species at risk on Pelee Island. Because of its well-delineated boundaries, Pelee Island provides a possible case study for future recovery actions of national and international initiatives.

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999.

Stewardship and the Reintroduction of Captive-Bred Burrowing Owls into British Columbia: A Feasible Option for Recovery?

Ernest E. Leupin
Ecoscape Biological Consulting
1405 Highway 97 South, Williams Lake, BC, V2G 2W3, Canada
leupin@direct.ca

Mike Mackintosh
City of Vancouver
2099 Beach Avenue
Vancouver, BC, V6G 1Z4, Canada

David J. Low
British Columbia Ministry
of Environment, Lands and Parks
1259 Dalhousie Drive
Kamloops, BC, V2C 5Z5, Canada

Dawn Brodie
Kamloops Wildlife Park
P.O. Box 698
Kamloops, BC, V2C 5L7, Canada

ABSTRACT

We present general aspects of a captive-breeding and release program developed for the burrowing owl (*Speotyto cunicularia*) in British Columbia. The program consists of 4 components that are essential for a successful reintroduction effort: breeding facilities; community involvement; private land stewardship; and field monitoring. Breeding facilities have become more finely tuned over time, and juvenile production is only limited by the space available at the facilities. The program has a strong community-based approach, which contributes to public awareness regarding the complexity and implications of endangered species recovery projects. This is achieved through education programs that emphasize the importance of managing the limited provincial grasslands through effective stewardship and hands-on involvement by the community. Field results indicate that 1-year-old captive-bred owls are capable of: 1) raising broods once released; 2) overwintering at or near release sites; 3) migrating south in winter, and; 4) returning to release sites the following spring. These results have provided a positive insight into the feasibility of reestablishing burrowing owl populations in British Columbia. The first phase of the program (1992–97) emphasized refining breeding strategies from the facilities and identifying critical habitat features necessary for owl survival and reproduction in the wild. Future activities will emphasize ecosystem restoration, taking into account ecological processes, (i.e., fire, grazing), and the resulting impact on grassland flora and fauna. This approach will be coupled with experimental testing of various release strategies.

Key words: burrowing owl, captive breeding, reintroduction, *Speotyto cunicularia*.

The burrowing owl (*Speotyto cunicularia*) is a small, migratory owl that inhabits grassland ecosystems. This owl differs from others in that it depends on burrows to reproduce and seek shelter. In most instances the owl occupies vacant badger (*Taxidea taxus*), coyote (*Canis latrans*), marmot (*Marmota* spp.), and ground squirrel (*Spermophilus* spp.) burrows. Decades of habitat alteration through urban and agricultural development, overgrazing, pest management, and natural disturbance suppression have resulted in a decline of several indigenous species, including the burrowing owl, in British Columbia's grasslands (Howie 1980). The burrowing owl was officially designated an endangered species in the province in 1980.

As early as 1980, the British Columbia Ministry of Environment, Lands and Parks launched a recovery program to restore the population of burrowing owls in British Columbia. One attempt initiated in 1986 involved the translocation of burrowing owl families from the Moses Lake area of Washington State to Oliver, B.C. in the Okanagan Valley. This particular project enjoyed limited success, and monitoring efforts were halted in 1989.

A second initiative involving a captive breeding and release program was initiated in the Thompson-Nicola region in 1989. The goal of this project is to establish self-sustaining populations of burrowing owls in at least 3 geographically

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

distinct locations in the southern interior of the province. Four components have been considered to be essential if the program is to be successful: breeding facilities; community involvement; private and public land stewardship; and effective field monitoring. In this paper we describe all of these aspects of the program and provide an initial evaluation of the release efforts.

BREEDING FACILITIES

There are currently 2 breeding facilities in British Columbia, geographically separated from each other to reduce the risk of catastrophic loss of the limited gene pool. Both facilities were established, and are operated, by private sector organizations. Construction of facilities has been accomplished with financial assistance from government, corporate, and private donations.

Kamloops Wildlife Park Breeding Centre

Constructed in 1989, Kamloops Wildlife Park is the main breeding and maintenance facility of the program. The pen design includes space for 8 breeding pairs. A central sheltering building (17 X 32 ft) contains separated nesting burrows that lead into individual exterior flyways for paired birds. A common flyway (10 X 100 ft) surrounds the individual enclosures and can be used by all birds outside of the breeding season. The building is equipped with power and heat for the winter season. Low-maintenance vegetation has been used throughout the display. Public viewing is restricted, with 1 side of the building being excluded from public view. Educational signage about the owl program is included for the park visitors, who can view the owls at a distance. With increases in breeding success in recent years, more space was needed for the juveniles, which required overwintering prior to their release in the spring. In 1996 a new juvenile pen (24 X 90 ft) was constructed immediately beside the breeding enclosures. The enclosure can hold up to 40 birds. The pen, which also contains 6 nesting chambers, can also serve as a second breeding facility.

San Rafael Aviaries Breeding Centre

From 1992 through 1997, a small facility was maintained at Stanley Park. When changes forced the removal of this enclosure, a new facility was constructed at San Rafael Aviaries near White Rock, B.C., with the support of the University of British Columbia Animal Science Department. The breeding centre consists of a large, outdoor aviary (50 X 50 ft), averaging 10 ft high, and divided into 3 sections to accommodate 3 breeding pairs. Two small buildings adjoining the flight cage contain nesting burrows for each enclosure; these connect to the outdoor flight pen via underground pipes. After the breeding season, partitions can be removed to allow communal use of the flyway space. Each breeding section contains additional nesting burrows, constructed above ground level and insulated with additional soil. The facility, completed in June 1998, was built by the Stanley Park Ecology Society Conservation Committee with funds raised from the Public Conservation Assistance Fund, and corporate and private donors.

PRIVATE LANDOWNER PARTICIPATION

The success of the reintroduction effort is totally dependent on the availability of suitable habitat. Some of the owls have been released to public land. However, with approximately 90% of grassland habitat in private ownership, the fate of the burrowing owl depends on a cooperative approach with the landowners. Landowners have made portions of their land available for establishment of nesting sites, altered traditional grazing regimes, and allowed access to monitor the released owls. The major property owners involved have been very supportive of the reintroduction concept. They are managing their resources in support of a healthy grassland ecosystem that is capable of supporting species that historically lived in the region, such as the burrowing owl and badger. These sound stewardship principles provide an opportunity for the reintroduction program to eventually be a success.

COMMUNITY PARTICIPATION

This program presents diverse opportunities for members of the public to be involved with the recovery effort. There is a wide spectrum of activities that can include people with a wide range of skills. Many age groups also become involved, from primary school students to senior citizens.

Maintenance of Breeding Facilities

Management of small populations requires care and attention to detail. Volunteers at Kamloops Wildlife Park and San Rafael Aviaries are involved in daily husbandry activities, record-keeping, maintenance of enclosures, and production of food supply.

Owl Sexing

Yearlings for release must be segregated by gender to prevent premature breeding attempts, and to manipulate successful, genetically diverse, breeding pairs in the field after release. Burrowing owls are sexually monomorphic and thus are difficult to sex in the hand. Researchers at the University College of the Cariboo Biological Sciences Department and Simon Fraser University Biotechnology Lab use DNA testing to determine gender. The test uses primers and PCR technology, which results in a 110-base-pair fragment that is present only in female samples.

Releases

When the owls reach 1 year of age, they are provided with pre-release training and are prepared for release to the wild. The recovery team has released birds at 8 locations throughout the Thompson-Nicola Valley. As a result of the low densities of burrowing mammals throughout the release area there are limited suitable nesting and shelter features available to the owls. Since 1992, volunteers from the Kamloops area and Lower Mainland meet twice yearly to install artificial burrows at sites pre-selected by the recovery team members. The work parties are usually scheduled for April to coincide with the release of the birds to the wild. A second trip takes place in the fall to maintain burrows, assess nest site use, and prepare additional sites for the following year. Nesting burrows are constructed from plastic pipe leading into an inverted bucket that serves as the nest chamber. A number of "shelter" burrows (without nest chambers) are also constructed in the vicinity of nest burrows. A series of burrow complexes will be built at a given release site to provide owl pairs with a variety of choices. Approximately 500 burrows have been constructed by volunteers across the release area.

Special Events

Volunteers also assist with the preparation of information brochures, logos, and other paraphernalia to raise funds and create awareness of the program.

Public Education and Interpretation

The charismatic nature of burrowing owls provides many opportunities for public education and interpretation. As a flagship species they can serve as ambassadors, representing features of the limited provincial grassland habitats. Both breeding facilities retain hand-raised owls, which are part of education programs related to endangered species and habitat protection. Over the past 10 years the Kamloops Wildlife Park education owl, Scout, has been introduced to more than 8,000 students and members of the community annually. The use of these live, captive-bred subjects in these presentations presents a powerful educational message with strong public appeal.

FIELD MONITORING

The successive releases of burrowing owls since 1991 have provided us with good, albeit mostly anecdotal, information that has allowed us to evaluate the feasibility of using a captive-breeding and release strategy to restore burrowing owl populations in British Columbia. To evaluate each release year separately would be lengthy and difficult. Instead, for the purpose of this paper we have opted to evaluate the program in terms of measures of success for all years combined. We have defined measures of success as the ability of the released owls to engage in behaviours typical of wild owls elsewhere, in the year of release (short-term), and in subsequent years (long-term).

SHORT-TERM MEASURES OF SUCCESS

Site Persistence and Owl Survival

Released yearlings showed high fidelity to release sites. Once released, most surviving owls remained at release sites. There were, however, exceptions: in 1 instance a female bird moved 4 km from the original release site to a second release site, where it paired with a resident male. This movement occurred after all other birds at the first site were depredated. Although abandonment of release sites was rare, owls often utilized structures other than the artificial burrows provided. Structures used included culvert pipes, spaces under abandoned buildings, discarded tires and, on one occasion, a natural burrow. Nevertheless, they eventually utilized the artificial burrows exclusively.

A total of 105 owls were released between 1992 and 1997. The average proportion of owl survival varied greatly between years ($\bar{\chi}$ = 0.63, range = 0.90–0.48), but not between release sites. Survival rates did not appear to be linked to gender; similar numbers of females and males persisted in any given year. The main source of mortality was predation. Observed instances of predation and recovered carcasses suggest that coyote and northern harrier (*Circus cyaneus*) were the main predatory species. Mortality rates in 1997 were reduced by 20% from the previous year. Initially, releases were carried out soon after the onset of spring (March–early April), which happened to coincide with major hawk migrations. It appeared that delaying the releases gave these "naïve" owls an opportunity to acclimate to the release sites by reducing their exposure to aerial predators.

The average annual mortality for captive-bred owls was similar to mortality rates elsewhere in their Canadian range. Schmutz et al. (1991) estimated annual survival in a small sample of burrowing owls in Alberta at 56%. In Saskatchewan, James et al. (1997) reported annual survival rates of 37–51%.

Dietary Habits

One concern regarding the release of captive animals is whether or not they will be able to secure food after release. We conducted analyses of pellets cast by the owls to determine their dietary habits. Pellets were collected throughout the spring, summer, and fall seasons. Pellet contents and food caches suggest that owls are able to secure food immediately after release. The main prey items were meadow vole (*Microtus pennsylvanicus*), deer mouse (*Peromyscus maniculatus*), northern pocket gopher (*Thomomys talpoides*), carrion beetle (*Silphidae* spp.), several species of ground beetle (*Carabidae* spp.), and spur-throated grasshopper (*Acrididae*

spp.). Other prey remnants found regularly in pellets included those of the Great Basin spadefoot toad (*Scaphiophus intermontanus*), western toad (*Bufo boreas*), western meadowlark (*Sturnella neglecta*), vesper sparrow (*Pooecetes gramineus*), mountain bluebird (*Sialia currucoides*), and western terrestrial garter snake (*Thamnophis elegans*). Small mammals comprised approximately 55% of the owls' diet and this proportion varied little throughout the year. In contrast, invertebrate prey intake reflected the seasonal availability of the various species. Owls consumed coleopterans almost exclusively during the spring and early summer, but gradually shifted to grasshoppers as the season progressed.

Published literature on burrowing owl food habits and the owls' response to the seasonal availability of prey (Maser et al. 1971, Grimm et al. 1985) suggests that the captive-bred owls' ability to secure prey is similar to that of wild owls.

Breeding Success and Productivity

Between 1994 and 1997, 31 young were produced from 12 nesting attempts. Mean clutch size was 5.6 (SD = 2.1). The number of successful nesting attempts was much lower. Five of the 12 attempts failed to produce a brood. Failure was attributed to the loss of 1 or both members of the pair as a result of predation. There were no instances where eggs were depredated. However, there was 1 incidence where an entire brood ($n = 5$) of 5-day-old chicks was cannibalized by the female, when the male failed to return. The mean brood size for pairs that successfully raised young was 4.1 (SD = 1.3). Evidence of renesting also occurred among released owls.

The observed brood sizes from released pairs are within the range of brood sizes observed in a long-term study in Alberta (Clayton 1997). In terms of nest failure rates, our average annual failure rate (44%) was relatively high compared to figures observed elsewhere. Haug (1985) reported nest failure rates of 21% in a study in Saskatchewan. Similarly, Haug and Churchward (1988) reported failure rates of 11% in Manitoba. Both studies attributed failures to avian and mammalian predators.

LONG-TERM MEASURES OF SUCCESS

Winter Migration and Residency

In general, surviving owls and their young disappeared from the release sites by early October. The location of their wintering grounds is unknown; however, a band recovery of a released owl in Ephrata, Washington in the winter of 1996 (approximately 500 km south from the original release site) suggests that owls are indeed moving long distances in a southerly direction within the Great Basin Grassland system of Oregon, Washington, and British Columbia.

Overwintering of owls was a rare but regular occurrence with released owls. Five of the 108 released owls remained at or near their release sites year-round. Two owls did so for 3 consecutive years. This behaviour was observed only in males. Overwintering owls did not appear to be adversely affected by winter conditions; 1 owl weighed 260 g in December, and another weighed 213 g in February (the average weight of a yearling owl prior to release at our facilities is 192 g). During this time the owls preyed almost exclusively on meadow voles and appeared to be active, except for days when temperatures dropped below 25°C or when burrows where covered with excessive snow.

Breeding Grounds Migration

Since 1993, 4 known released owls have returned to their original release sites after leaving them for the winter months. In all instances the returning owls were males released the previous year. There is no evidence that any females have returned to release sites. However, on 1 occasion a wild female was observed with a returning male. The wild female successfully raised 5 young.

DISCUSSION

The burrowing owl program in British Columbia presents an important model for recovery projects; one that reflects community diversity that will be essential for the success of the program. A cooperative relationship exists between government and private organizations that have a direct role to play. The large landowners involved have supported the efforts on their property. The grassroots nature of the program has also provided some natural opportunities for public involvement. Many of the activities are readily accessible to people who want to become environmental stewards in a direct manner. In the Kamloops area, through education programming, there is a strong community identification with the burrowing owl story. Finally, for all that participate in the program there is the sobering reminder that the business of recovery of an endangered species is a daunting task requiring considerable effort.

Our observations give grounds to be optimistic about an eventual reestablishment of burrowing owl populations in British Columbia. However, productivity and mortality rates of the released owls are similar to those observed in declining populations elsewhere in Canada (Wellicome 1997). Therefore, our immediate efforts will focus on habitat management strategies that will improve productivity and the number of returning owls, and reduce mortality rates.

In a natural situation in British Columbia the bulk of the owl's diet is made up of small mammals (Leupin and Low 1996). Wellicome (1997) showed that supplemental feeding to pairs during the nestling period resulted in increased production of young when compared to unfed pairs. Therefore, increases in prey base availability should increase owl productivity. Current grazing regimes in British Columbia are for the most part inadequate, in that little residual cover is

left behind to support high densities of prey species. Burrowing owls prefer grazed areas with little vegetation to nest in (Coulombe 1971, Rich 1996); this habitat type is unsuitable for most of their vertebrate prey species. Small mammals, particularly meadow voles, are associated with riparian areas and dense cover (Dickinson et al. 1994). Therefore, in order to be successful, the owls need a combination of habitats that provide both nest sites and foraging areas in close proximity to each other. We intend to work closely with landowners to implement grazing regimes that promote such habitat heterogeneity, and thus provide suitable habitat to both burrowing owls and their prey species.

The breeding of owls at our facilities has become more finely tuned over time, and we now have the potential to produce >100 individuals annually. This would allow us to conduct group releases of at least 25 pairs at 2 separate release sites. Group releases in these numbers should reduce mortality rates, increase the number of returning individuals, and improve productivity at release sites (A. Sinclair, Department of Zoology, University of British Columbia, 1997, pers. comm.).

Finally, burrow availability has been cited as a key factor contributing to the decline of burrowing owls in British Columbia (Howie 1980). The shortage of burrows has come about from a reduction in fossorial mammal populations. Although artificial burrows are an effective short-term enhancement technique, they should not be considered an ultimate solution (Bryant 1990). Yellow-bellied marmot (*Marmota flaviventris*) and badger are 2 species that still persist in British Columbia's grasslands. In future years we will concentrate on attempting to restore populations of these burrowing mammals into suitable habitats, which should in turn provide a natural source of burrows for the owl.

The recovery team has experienced small, measurable gains in the effort to return sustainable numbers of owls to the southern interior of the province. There are no quick solutions. Patience, knowledge, and perseverance will be required if the burrowing owl is to return in sufficient numbers over a wide enough area so that human involvement will no longer be necessary.

ACKNOWLEDGEMENTS

Funding for this project has been provided by the British Columbia Habitat Conservation Trust Fund. Additional funding has been kindly provided by: Canada Trust Friends of the Environment Foundation; Public Conservation Assistance Fund; Trans Mountain Pipeline Ltd.; Shell Environmental Fund; Petro Canada; Frederick Metcalf Charitable Foundation; Imperial Oil; and Molson Industries. We thank all volunteers and landowners, especially the Kamloops Wildlife Park staff and volunteers, the Stanley Park Ecology Society Conservation Committee, and Guichon Ranch, for without their support and involvement this project would not have been possible. The British Columbia Conservation Foundation provides administrative support and aids greatly with fundraising. Finally, we thank M. Chutter for valuable comments on an earlier version of this manuscript.

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Recovery of Wood Bison in British Columbia

William L. Harper
Osiris Wildlife Consulting
4399 Shore Way, Victoria, BC, V8N 3V1, Canada
bill.harper@home.com

C. Cormack Gates
Faculty of Environmental Design
University of Calgary, Calgary, AB, T2N 1N4, Canada

ABSTRACT

Wood bison (*Bison bison athabascae*) were nearly eliminated by uncontrolled hunting during the fur trade in the late 1800s. In British Columbia, the last confirmed wood bison was shot in 1906. Canadian populations began to recover in the early 1900s, but the genetic isolation of the original population was transgressed with the introduction of plains bison (*Bison bison bison*) into Wood Buffalo National Park (WBNP) in the late 1920s. Bovine tuberculosis and brucellosis were also introduced with the plains bison. Wood bison salvaged in 1965 from WBNP were used to establish a disease-free, national captive-breeding herd that has since provided stock for 6 free-ranging populations. Recent studies confirm the descendant herds resemble wood bison, despite the earlier hybridization event. One goal of the Canadian wood bison recovery program is the establishment of at least 4 herds of more than 400 animals each, with at least 1 of them in British Columbia. Two herds outside of British Columbia have reached the minimum size threshold, and the total number of animals in specific disease-free, free-ranging populations is now over 2,800, although tuberculosis- and brucellosis-infected herds in and around WBNP remain a significant threat. An estimated 70 wood bison now occur in British Columbia. Large areas of unoccupied wood bison habitat exist in northeastern British Columbia. In the late 1980s, British Columbia established a "bison control area" to prevent hybridization between wood bison and plains bison. Forty-nine wood bison were translocated to the Liard River area in 1995. The goals of the updated Management Plan for Wood Bison in British Columbia are to continue to reestablish herds through translocation (e.g., Etthithun Lake), to maintain their separation from plains bison, to maintain their disease-free status, and to allow populations to increase to a level sufficient to support nonconsumptive and consumptive use. The biggest obstacle to successful implementation of the plan is the presence of game-farmed and free-ranging plains bison within the historic range of wood bison. Developing and maintaining public support for recovery efforts is essential.

Key words: *Bison bison athabascae*, *Bison bison bison*, game-farming, Liard River, plains bison, recovery, reintroduction, translocation, wood bison.

Prior to European settlement, wood bison (*Bison bison athabascae*) were widely distributed over a large area of the western boreal forest in Canada (Fig. 1), including northern British Columbia, although they were unevenly distributed and never as numerous as plains bison (*Bison bison bison*; Gates et al. 1992). Soper (1941) roughly estimated wood bison numbers at 168,000 in the early 1800s. Like the plains bison, wood bison were nearly eliminated by uncontrolled hunting for the fur trade during the late 1800s. Populations declined most rapidly after 1860, and reached an estimated low of 250 between 1896 and 1900 (Soper 1941). Although legislation designed to protect bison was passed in Canada in 1877 and 1893, enforcement was not effective until after the turn of the century. It was only when the Northwest Mounted Police were given responsibility for enforcing the Buffalo Protection Act in 1897, and formal patrols began in 1907, that wood bison populations began to recover south of Great Slave Lake (Gates et al. 1992). In British Columbia, the last bison from the primordial population was shot near Fort St. John in 1906 (MacGregor 1952). They remained extirpated until recovery actions were implemented in 1980.

Between 1925 and 1928, more than 6,000 plains bison were translocated to historic wood bison range at Wood Buffalo National Park (WBNP) in order to relieve overcrowded conditions at Buffalo Park in Wainwright, Alberta (Wood Bison Recovery Team 1987, Gates et al. 1992). Two cattle diseases, brucellosis and tuberculosis, were introduced with the plains bison (Tessaro et al. 1992). All contemporary stocks of wood bison, including those in and originating from WBNP, are derived from the hybridization of wood and plains bison (van Zyll de Jong 1986, Wilson and Strobeck 1998). However, the influence of the wood bison genome remains strongly expressed in the phenotype of WBNP bison and descendant populations (van Zyll de Jong et al. 1995).

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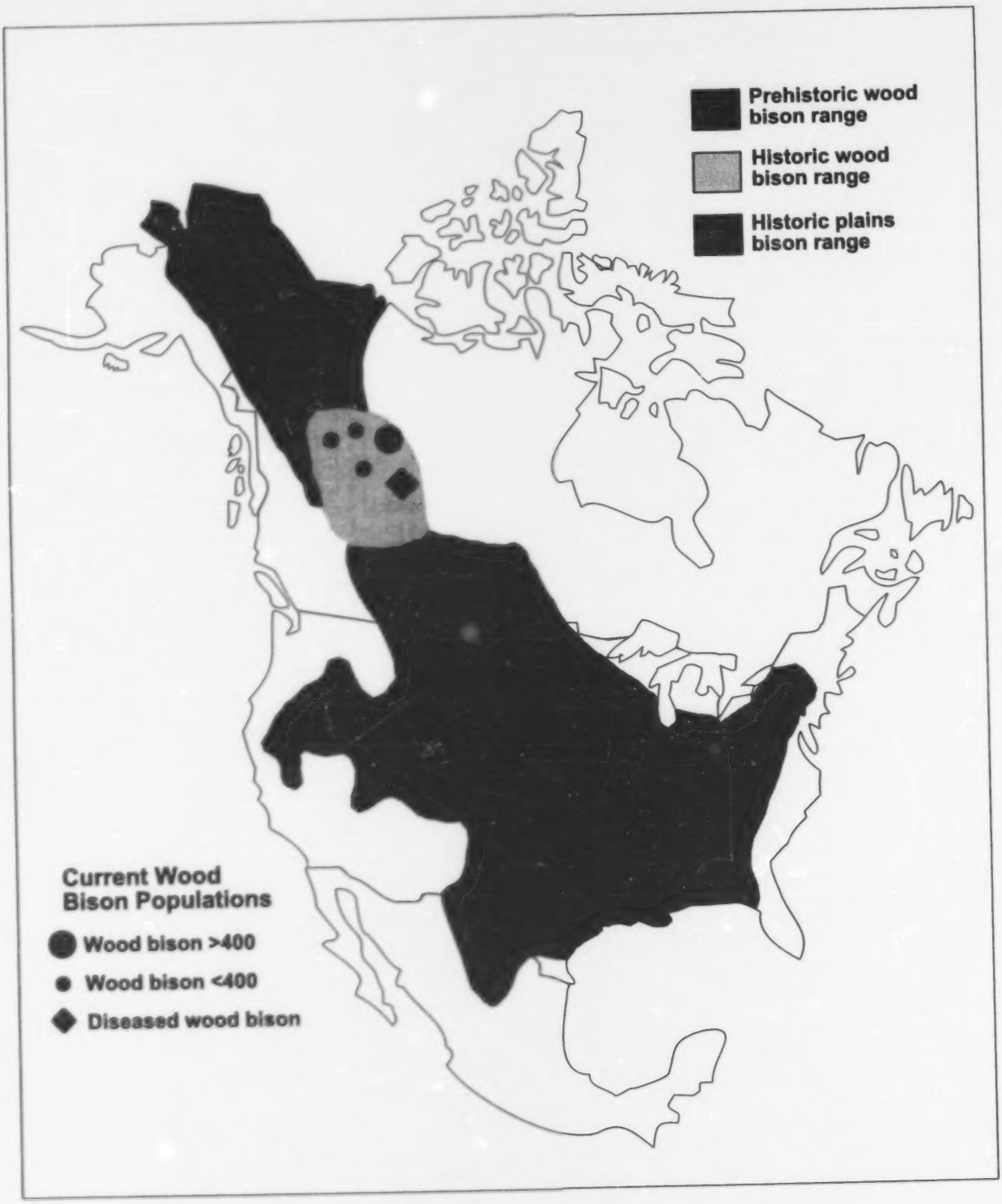

Figure 1. Historic distribution of bison in North America.

Speculation starting in 1959 about the existence of an isolated northern population of pure wood bison (Banfield and Novakowski 1960) resulted in the capture and translocation of bison from the Nyarling River area of WBNP in 1963. Eighteen wood bison were successfully translocated to the Mackenzie Bison Sanctuary (MBS) near Fort Providence, Northwest Territories. A second translocation occurred in 1965, when 21 wood bison were moved to a fenced enclosure at Elk Island National Park (EINP) in central Alberta. Another salvage project was conducted during 1996–98, when 62 neonates were captured from the Slave River Lowlands in the Northwest Territories and transferred to a captive breeding facility in Fort Resolution (Gates et al. 1998).

In the mid 1970s, the Committee on the Status of Endangered Wildlife in Canada (COSEWiC) designated wood bison as Endangered, and a recovery program was established with representatives from federal, provincial, and territorial wildlife agencies and Parks Canada. Recovery efforts accelerated through the 1980s with the appointment of the Wood Bison Recovery Team (WBRT) and the down-listing of wood bison to Threatened (Wood Bison Recovery Team 1987). The most controversial problem then, as now, was the issue of what to do about the brucellosis- and tuberculosis-infected bison in and around WBNP. Various actions have been proposed, including the slaughter and replacement of all diseased bison inside and outside of the park (EAP 1990), and conservation and recovery of infected herds through a salvage and captive breeding program (Gates et al. 1998).

A bison management plan for British Columbia (B.C. Wildlife Branch 1991) was prepared in the late 1980s. The plan proposed reintroducing wood bison north of Fort Nelson, preventing further hybridization with plains bison by designating a "bison control area," and not permitting commercial operations of game-farmed plains bison north of 57°N latitude. Since the initial plan was written, many biological and socio-political factors that affect wood bison conservation efforts have changed, making it necessary to update the 1991 plan.

The currently proposed *Management Plan for Wood Bison in British Columbia* (Harper et al. 1999) is designed to support key elements of *A Wildlife Policy for Canada* (Wildlife Ministers' Council 1990), the *Canadian Biodiversity Strategy* (Biodiversity Working Group 1994), and the *British Columbia Provincial Wildlife Strategy* (B.C. Wildlife Branch 1994). Most importantly, the proposed *Management Plan for Wood Bison in British Columbia* is a key component of the *National Recovery Plan for Wood Bison* (Gates et al. in prog.). In concert with other provincial and territorial bison plans and programs (e.g., Reynolds et al. 1985, Government of Northwest Territories 1987, Government of Yukon Territory 1998), the British Columbia management plan is designed to support the goals and objectives of the National Recovery Program.

CONSERVATION STATUS

The International Union for the Conservation of Nature (IUCN) includes North American bison on their list of Lower Risk–Conservation Dependent species (Baillie and Groombridge 1996). The Convention on the International Trade of Endangered Species of Fauna and Flora (CITES) lists wood bison in Appendix II. The United States listed wood bison as "Endangered in Canada" under their federal Endangered Species Act in 1970 and has not updated that listing since.

COSEWIC lists wood bison as Threatened. The British Columbia Conservation Data Centre categorizes wood bison as S1 (critically imperiled because of extreme rarity) under the Nature Conservancy ranking system. The British Columbia Ministry of Environment, Lands and Parks includes wood bison on its Red List.

CURRENT DISTRIBUTION OF WOOD BISON

Canada

Free-roaming wood bison occur only in Canada at present. The 1998 population estimate for free-roaming, disease-free wood bison in Canada is 2,800 animals in 6 geographically separate herds (Gates et al. in prog.). There are also 860 wood bison in 4 co-managed, captive-breeding herds, and 500–700 animals on 45–60 private, commercial game farms. There are also approximately 2,300 wood bison infected with bovine tuberculosis and brucellosis in and around WBNP.

British Columbia

Wood bison returned to British Columbia in 1980 as dispersing animals from the Nahanni Butte reintroduction in the Northwest Territories, and in 1995 and 1998 as the result of direct reintroductions of animals bred at EINP. All wood bison in British Columbia are descendants of the 21 Nyarling River animals captured near Needle Lake in WBNP in 1965, which were relocated to EINP (Wood Bison Recovery Team 1987). Reintroduced wood bison currently occur in 2 separate locations: Beaver River; and the Nordquist Flats area near the Yukon/Northwest Territories border (Fig. 2).

Nordquist Flats Herd

The first reintroduction of wood bison to British Columbia took place in March 1995. Forty-nine animals (4 adult males, 5 adult females, 5 yearling males, 10 yearling females, 10 male calves, and 15 female calves) were transferred from EINP to Aline Lake in the Nordquist Flats area of the Liard River valley (Fig. 2). This translocation site is approximately 80 km from the Nahanni Butte bison herd at the mouth of the Beaver River. A vegetation inventory of the translocation site in 1988 indicated large areas with upland grass

production between 500 and 900 kg/ha, and smaller wetland areas with sedge production of approximately 6,300 kg/ha (Elliott 1989).

Beaver River Herd

As of 1998, the herd reintroduced at Nahanni Butte had expanded its range upstream along Liard River to a point approximately 40 km south into British Columbia, near the mouth of the Beaver River (Fig. 2). An estimated 30 wood bison occur along the Liard River from the mouth of the Beaver River to the border with the Northwest Territories. Approximately half of these are animals that had dispersed, at 1 point, as far south as Fort Nelson after being released near Nahanni Butte in April 1998 (J. Nishi, Northwest Territories Resources, Wildlife and Economic Development, pers. comm.).

Etthithun Lake Herd

In March 1996, 18 wood bison that had been maintained for 2 years at Northern Lights College in Dawson Creek, B.C. for captive studies were released to the wild near the headwaters of the Etthithun River near the Alberta border (Fig. 2). Habitat primarily consists of a mixture of domestic and native grasses that occur along road allowances, seismic lines, pipelines, and well sites. Fifteen bison were observed in January 1997; by March 1997, 3 of the 15 had been killed in collisions with industrial road traffic (Churchill and Maundrell 1998). In the summer of 1997 the herd spent an increasing amount of time up to 100 km south, on the fringe of agricultural development. They eventually joined a small herd of feral commercial bison that had escaped from a ranch. The mixed herd of 26 plains and wood bison was captured later that summer and ultimately sold to a private ranch 120 km south of the original release site.

In March 1999, 19 wood bison from EINP were moved to a recently constructed captive breeding facility in the Etthithun Lake area. The herd will be held for several years to habituate the bison before they are released to the wild.

Hay River

Wood bison from the Hay-Zama reintroduction in Alberta occasionally wander into the Hay River drainage in northeastern

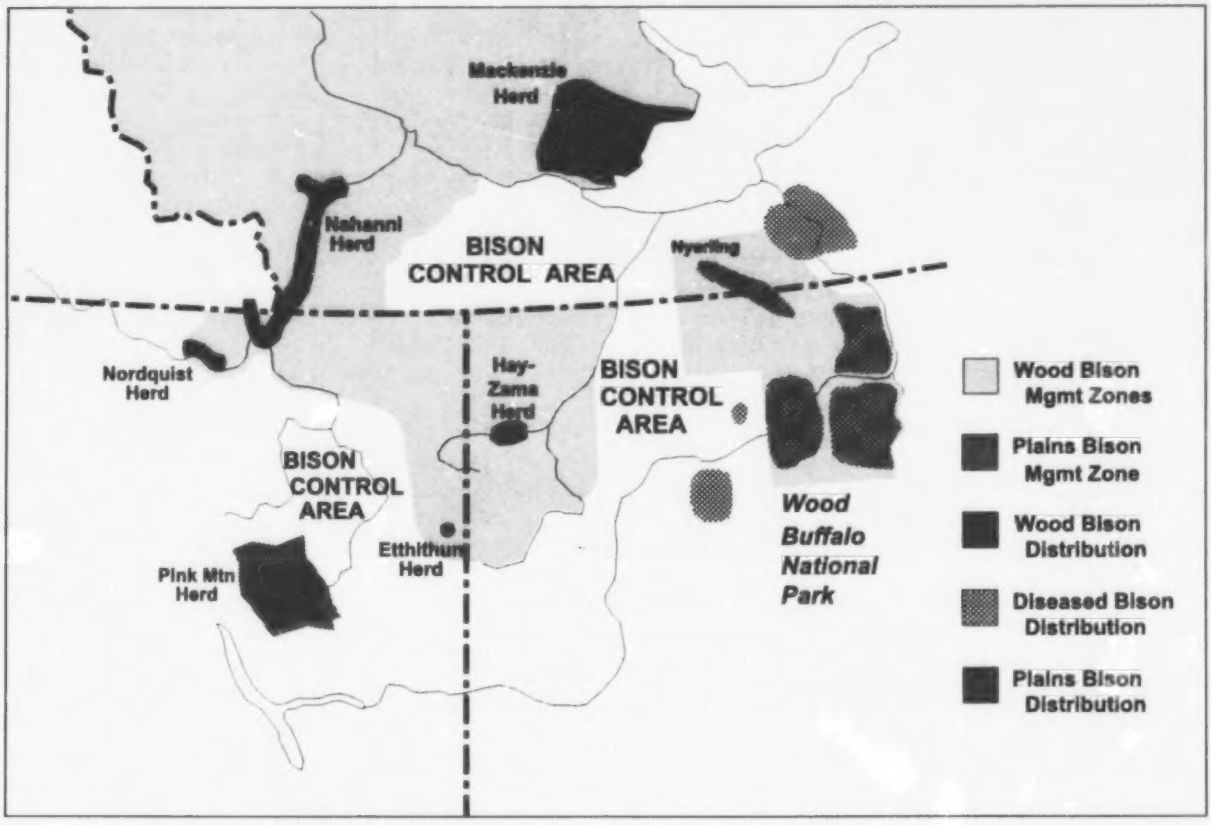

Figure 2. Distribution of management zones and free-roaming herds of bison.

British Columbia (Fig. 2). As the Alberta population grows, it is likely it will eventually expand to establish a more permanent population in this part of British Columbia.

CURRENT DISTRIBUTION OF PLAINS BISON IN BRITISH COLUMBIA

Plains bison accidentally introduced in the Pink Mountain–Sikanni Chief River area in 1971 have increased in distribution and abundance, particularly over the past 15 years. The herd is largely confined to grass/sedge meadows in the upper Sikanni River and Halfway River valleys (Fig. 2). In 1990 an estimated 500 animals ranged over an area of 500 km^2. The current population ranges over an area of similar size, but it has increased to an estimated 1,400 animals (J. Elliott, B.C. Ministry of Environment, Lands and Parks, pers. comm.). Potential exists for forage competition between free-ranging plains bison and wood bison if the ranges of the two would overlap, but the policy of restricting each subspecies to mutually exclusive ranges to prevent hybridization means this should never occur.

SOCIO-POLITICAL CONSIDERATIONS

Legal

British Columbia Wildlife Act

Bison are designated as "wildlife" and "big game" under the British Columbia Wildlife Act. This designation vests ownership of all wild bison with the provincial government, and protects them from deliberate killing, capturing, or personal possession, except as authorized by permit. Once a game-farmed bison escapes captivity, or is released or abandoned, the provincial government acquires ownership of that animal. Wildlife Act permits are required for the movement of bison across the borders of British Columbia, both to import live animals, and to export bison or bison products.

British Columbia Game Farm Act

The British Columbia Game Farm Act defines "game" as fallow deer, bison, and reindeer. Under this act, a licence or licence renewal may be issued by the Ministry of Agriculture and Food (MAF) to a person to raise a type of game for agricultural purposes. Provisions of the Wildlife Act with respect to escaped animals do not apply to licensed game if the licensee recaptures the animal within 30 days. The Game Farm Act also includes provisions enabling Cabinet to make regulations specifying regions of British Columbia in which a licence for a specified type of game must not be issued. Regulations may also establish different terms and conditions for different licensees based on geographical area. There are currently no regulations that specify regions or conditions that pertain to licences for bison game farms.

Commercial Bison Farming

Commercial bison farming is an established agricultural industry, and most bison in North America are now managed as livestock. Commercial production is expected to increase by at least 25% annually until 2005 (Marchello 1998) reaching over one-half million behind fence by the year 2000 (Hudson 1998). Plains bison dominate commercial production of bison, since commercial production of wood bison did not begin until 1989. In northeastern British Columbia there are currently 49 producers of commercial bison registered under the British Columbia Game Farm Act. Problems can arise if farmed animals become diseased, or if escaped animals are allowed to mix with free-roaming herds. The Peace Country Bison Association lists 250 members in northern Alberta and British Columbia (D. Patten, Canadian Bison Association, Bezanson, AB, pers. comm.).

Disease Considerations

Three significant infectious diseases (tuberculosis, brucellosis, and anthrax) are present in wild populations of wood bison in the Northwest Territories and northern Alberta. Bovine tuberculosis and brucellosis were introduced to WBNP by plains bison in the late 1920s. The population of bison in WBNP has declined to 21% of its former level in the past 27 years (Joly et al. 1998). This decline is attributed to chronic tuberculosis and brucellosis infections and a resultant loss of productivity and recruitment. At this point, none of these bovine diseases have been diagnosed in wild ungulate populations in British Columbia. "Bison control areas" established by Alberta and the Northwest Territories (Fig. 3) will hopefully help prevent spread of these diseases from WBNP.

Domestic bison ranches are also a potential threat to the disease-free status of recovering populations of free-ranging wood bison. Various livestock diseases, if present in game-farmed animals, could spread to wild populations by contact through fences, or from escaped animals.

Genetic Considerations

Interbreeding with Plains Bison

Escapes and unauthorized translocation of commercial bison are substantial threats to the genetic isolation and recovery of free-ranging wood bison in Canada (Gates et al. in prog.). The rapid expansion of the commercial bison ranching industry in British Columbia has resulted in an increased number of escapes and rising potential for additional hybridization with free-ranging wood bison. The size and strength of the bison make it a difficult animal to permanently contain behind fencing. Policies within MAF are in place to report escaped animals, which become the property of the government if not recaptured within 30 days. Problems can exist with effective enforcement of policies to round up or dispose of feral domestic bison abandoned by ranchers.

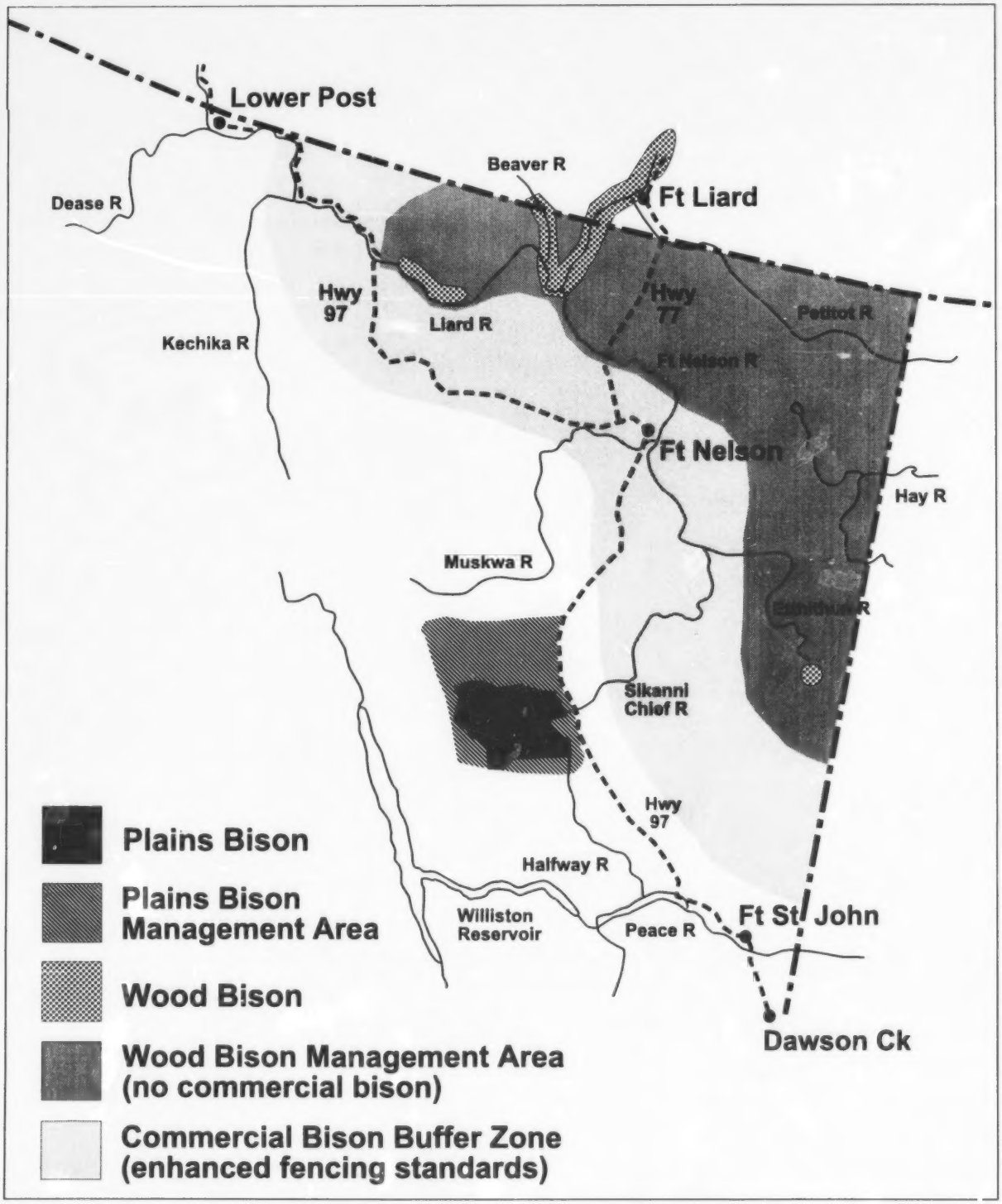

Figure 3. Proposed management zones for bison in British Columbia.

Improving Genetic Diversity

Microsatellite DNA indicates that wood bison from EINP and MBS animals are less genetically variable than subpopulations from WBNP, confirming suspicions regarding the effect of small founding populations (Wilson and Strobeck 1998). The genetic diversity of reintroduced populations originating from EINP could be augmented by introduction of disease-free animals from WBNP, the Slave River Lowlands, or the Mackenzie bison population.

Bison-Vehicle Collisions

Due to the large size of bison, collisions with vehicles are a substantial public safety and property damage issue. Wood bison in the Yukon Territory have caused traffic hazards along the Alaska Highway, with 8 killed in collisions, 5 killed as "problem" wildlife, and 36 captured and relocated to a game farm. The Yukon government has a "bison-free" policy in the vicinity of the Alaska Highway involving deterrence, capture, and, ultimately, disposal of "problem" animals (Government of Yukon Territory 1998). Three of 15 animals reintroduced to Etthithun Lake in 1996 were killed in collisions with industrial road traffic during the first winter (Churchill and Maundrell 1998).

Hunting

Hunter harvest can be used to achieve a number of objectives. It can be used to control the distribution of animals in order to implement "bison control areas" designed to prevent disease transmission or interbreeding with plains bison. It can also be used to control population size and density if the objective is to reduce the impact of bison grazing on other components of the ecosystem. The policy of the British Columbia government is that Red-listed species such as the wood bison are managed for recovery and not hunter harvest. Once populations have built up to a level that supports a harvestable surplus, hunting can also provide a source of food, recreation, and input to the provincial economy.

Land Use Considerations

Urban and agricultural development is the land use that is least compatible with wood bison population objectives. Wood bison are rarely tolerated near human habitations. Alienation of habitat for agricultural production is even more of a threat because of the large areas involved. Agricultural development in the Fort St. John and Fort Nelson areas has greatly reduced bison habitats, and continuing expansion of agriculture in the north will further limit the ability to meet population recovery objectives. Geographic expansion of the bison ranching industry may also limit the amount of land available for establishing free-ranging wood bison.

Forestry is increasingly becoming a resource use within the historic range of wood bison. There are substantial opportunities to maintain or increase available habitats for wood bison through forest management because of the wood bison's preference for open habitats. Bison are adapted to forage in open grassy environments and may benefit from certain alterations to the environment associated with oil and gas exploration (e.g., seismic lines).

Intensive habitat management can be used to improve foraging opportunities for wood bison. Suppression of wildfire has led to an alteration of natural fire regimes that encourages shrubs and trees at the expense of bison meadow habitats (Gates et al. in prog.). Prescribed burning has been used to replace these natural fire regimes in the Northwest Territories, with up to 27,000 ha treated in some years (Chowns et al. 1998). On a smaller scale in British Columbia, prescribed fire has been used to improve habitat conditions on bison range in the vicinity of the Nordquist Flats.

PROPOSED MANAGEMENT PLAN FOR BRITISH COLUMBIA

Although wood bison are no longer in immediate threat of extinction in Canada, further work is required to ensure their long-term survival (Gates et al. in prog.). The provincial management plan (Harper et al. 1999) is designed to recover the species in British Columbia, down-list wood bison from the Red List, contribute to the goals of the national recovery plan, and eventually build up populations to levels that could support sustainable human use. The management plan outlines the goals, objectives, and specific actions required to recover wood bison populations in British Columbia, recognizing that it is no longer feasible to return wood bison to their original distribution or abundance. There are 4 basic principles on which the management plan is based:

1. The recovery of wood bison in British Columbia will be based on the establishment and maintenance of wild, free-roaming populations in suitable habitats, within their original geographic range in British Columbia.
2. All efforts will be made to preserve the genetic isolation and diversity of wood bison and maintain their current genetic distinctiveness from plains bison.
3. In order to provide for continued natural evolution of wood bison, the normal interaction between free-roaming wood bison and their native environment will be maintained.
4. To the greatest extent possible, the management plan will support the goals of the national recovery program and the objectives of wood bison management plans in adjacent jurisdictions.

Goals and Objectives of the British Columbia Management Plan

Goal 1

Reestablish a viable population of at least 1,000 disease-free, free-roaming wood bison within their former range in a

designated area in British Columbia (Fig. 3). More than 1 or 2 populations are required to reduce the risk of random events exterminating the British Columbia population. Ultimately the goal would be to have all populations in British Columbia interconnected as 1 metapopulation.

Goal 1 Objectives

(a) Ensure wood bison are reestablished within their historic range, and in areas with sufficient habitat to support viable populations; (b) evaluate high-priority translocation sites based on a detailed evaluation of habitat and potential land-use conflicts; (c) translocate wood bison to at least 3 geographically separated sites with sufficient suitable habitat to support population objectives; (d) augment existing free-roaming populations when feasible and necessary to achieve minimum viable population or improve genetic diversity; (e) manage each reintroduced population with adjacent jurisdictions to increase to at least 400 animals; and (f) manage habitats of established populations to achieve population recovery objectives.

Goal 2

Maintain the genetic isolation of free-ranging wood bison in British Columbia. Hybridization with plains bison 70 years ago threatened the wood bison as a distinct subspecies and remains 1 of the greatest threats to provincial and national recovery programs.

Goal 2 Objectives

(a) Prevent interbreeding between free-ranging plains bison at Pink Mountain and populations of reintroduced wood bison; (b) prevent interbreeding between game-farmed commercial bison and populations of reintroduced wood bison; and (c) evaluate the potential for improving the genetic diversity of existing free-roaming populations. A management zone will be established in which limitations will be imposed on bison ranching activities (Fig. 3).

Goal 3

Maintain the disease-free status of free-ranging wood bison in British Columbia. The presence of introduced bovine diseases in bison herds in and around WBNP is the single greatest obstacle to the recovery of wood bison in Canada. All efforts must be made to prevent these diseases from entering British Columbia.

Goal 3 Objectives

(a) Ensure all free-roaming and domestic bison that enter British Columbia are free of serious diseases such as anthrax, tuberculosis, and brucellosis; and (b) monitor the disease status of free-roaming and domestic bison.

Goal 4

Allow free-roaming, disease-free wood bison populations to increase to a level that would allow for long-term, sustainable human use. There is significant potential for ecotourism, wildlife viewing, resident hunting, nonresident guide-outfitting, and native subsistence use. Achieving population levels that allow human use of wood bison is consistent with, and supports, conservation objectives.

Goal 4 Objectives

(a) Monitor changes in population distribution, size, and demographics; (b) manage both habitat and populations to achieve overall population objectives, prevent overgrazing, and maintain natural ecosystem function; (c) optimize opportunities for residents and visitors to view free-roaming wood bison; and (d) eventually adopt conservative hunting seasons consistent with the British Columbia Wildlife Harvest Strategy.

Goal 5

Work cooperatively with public groups, rural communities, aboriginal peoples, and adjacent jurisdictions to develop mutually agreeable objectives for establishing and maintaining healthy, free-roaming wood bison herds in British Columbia. Conservation objectives are easier to achieve if local communities have a direct stake in reestablishing and maintaining viable populations of wood bison in their area. Native peoples within the historic range of wood bison have a long history and cultural attachment to this species.

Goal 5 Objectives

(a) Involve government agencies, stakeholders, public groups, rural communities, and native groups in development of reintroduction plans; (b) consider the development of long-term cooperative management programs with rural communities and native groups for reestablishing healthy, free-roaming wood bison herds; and (c) consult neighbouring jurisdictions during the development of reintroduction plans and wood bison policy changes.

ACKNOWLEDGEMENTS

We thank H. Schwantje, J. Elliott, I. Hatter, D. Fraser, M. Hoefs, and J. Nishi for comments on an early draft of the provincial wood bison plan. G. Diemert, D. Moyles, A. Ackerman, and M. Wetzstein also participated in discussions on how to address commercial bison farming issues.

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VGI Vision Foundation

M.G. Shepard
VGI Vision Foundation,
Suite 211 - 5325 Cordova Bay Rd., Victoria, BC, V8Y 2L3, Canada
mgs@vgivision.com

ABSTRACT

The VGI Vision Foundation is a non-profit organization designed to encourage the gathering and free flow of environmental data. Through web-based technology the Foundation is building up an environmental database to provide data for resource managers, industry, government agencies and amateur naturalist communities. A major function of the Foundation is to encourage volunteer participation in research and monitoring activities. Currently the Foundation is coordinating the operation of the Rocky Point Migration Monitoring Project. The poster presentation highlights the objectives of the organization and displays results from the Rocky Point project.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk. Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Outdoor Discoveries Interpretive Services

R. Van Ee

Outdoor Discoveries Interpretive Services
8283 Silver Star Rd., Vernon, BC, V1B 3M8, Canada
outdoors@bcgrizzly.com

ABSTRACT

Well-designed interpretive signage informs and educates the public. It is increasingly being used as a management tool within the process of fulfilling other management objectives. Outdoor Discoveries has been contracted by the B.C. Ministry of Forests, forestry companies and non profit organizations to develop interpretive signage as a part of a management requirement. Examples include: 1) BATS IN THE FOREST sign, for the Ministry of Forests, Vernon (this sign explains bat habitat research in progress in a demonstration forest); 2) UPPER GRAYSTOKES ORIENTATION sign, for the Ministry of Forests, Penticton (this sign was produced to precede a formal road closure in sensitive subalpine wetland habitat that was being destroyed by 4-wheel drive vehicles); and 3) BEETLE KILL HARVESTING sign, for Riverside Forest Products, Lumby (this sign was produced to explain beetle kill harvesting of Doulglas-fir in a watershed where public concern was high). These 3 examples will be featured to explain the use of well-designed interpretive signage as an effective management tool. Actual examples of signs will be displayed. Photo samples with labels will also be displayed on the table.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 526pp.

Hyperdiversity and Risks in Environmental Education

Richard Kool

BC Parks, British Columbia Ministry of Environment, Lands and Parks
P.O. Box 9398, STN PROV GOVT Victoria, BC, V8W 9M9, Canada
kool@pacificcoast.net

ABSTRACT

My conclusions are: Environmental Education (EE) is worth doing because it is good for teachers and for students; we have to be careful not to fragment ourselves into groups with little chance of succeeding; we have to network, to connect, to collaborate, to manage our egos, to work together; we have to celebrate diversity without becoming so diverse that we cannot sustain our efforts; we have to understand that the world is changing around us—our assumptions about how things work are undergoing a radical shift and we had best keep up with these things; and being an innovator is never easy, but it may be fun.

Key words: education, environmental education, teaching.

I attended a conference last fall where the opening speaker was Dr. Arun Gandhi telling a story of his grandfather, the great political and spiritual leader of India. The Mahatma lived in an ashram in India, and for about 2 years, when he was between 12 and 14, young Arun lived with his grandfather. There was a family living in the ashram with a young son who loved to eat sweets. This boy would eat too many sweets, steal sugar from other homes, and was breaking out in boils because of all the sugar he was eating. The parents tried to discipline their son, but with no success. In desperation, they brought the boy to the Mahatma, explained their problem, and asked him to talk to the boy and tell him to stop eating so many sweets. But instead, Gandhi told the parents to go away for 15 days and then come back, then he would talk to the boy.

On the appointed day, the family returned to Gandhi's house. The Mahatma went over to the boy, whispered into the boy's ear, the boy nodded, and all was done. The boy agreed to stop eating sweets.

The parents were amazed, and asked Gandhi what did he say, what was the magic that he put into their son's ear, and why couldn't he have said what he said on the day of their first visit. Arun Gandhi couldn't hear what his grandfather originally said to the boy, but he did hear the old man's response to the parents' question. Gandhi told them that first, before he could talk to their son and ask him to do something that would be very hard for the child to do, Gandhi had to do a hard thing—stop eating sugar—himself. And he knew it would take him a few weeks to do this.

Are we willing to do the hard things that we, as educators and communicators, expect others to do?

There are a lot of similarities between endangered species and an endangered form of education: environmental education (EE).

EE is a product of an evolution that began in the early part of this century with nature studies in schools and communities, itself an outgrowth of the 19th century interest in natural history driven, in part, by the 18th century interest in natural Theology, the study of God as He reveals Himself in His continuing act of creation. From Nature Study in the middle of this century, educational efforts metamorphosed into Conservation Education. Still pretty focused on nature study, conservation education tried to help students and teachers understand the importance of saving and preserving natural areas. Conservation Education further evolved, in part due to Rachel Carson's influential *Silent Spring*, through the 1960s. By the early 1970s, natural selection was again at work, selecting against the old conservation education folks and in favour of what was now being called Environmental Education, an educational approach that involved more of an integration of natural and built environment, more of an action-orientation towards the environment and education, and more consideration of the relationships between environment and economy.

There's allopatric speciation as well. We see this in the development of small and always endangered species of educators differentiated by habitat type. As you all know, the most common way for species to split is when a population becomes geographically isolated in 2 separate areas, allowing the 2 groups to follow different evolutionary paths. They are likely to differentiate for 2 reasons:

1. Different geographic regions are likely to have different selective pressures. Temperature, rainfall, predators, and competitors are likely to differ between 2 areas. Thus, over

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time, the 2 populations will differentiate.

2. Even if the environments are not very different, the populations may differentiate through genetic drift, because different mutations and genetic combinations occur by chance in each.

It's always been important for us to ensure that the marine and aquatic educators, the forest educators, the big-hairy-mammal educators, the bird educators—all those incipient species of EE—keep connected, keeping us all in a subspecies or variety level of relationship. Too much diversity here may not be a good thing.

But keeping in the evolutionary model mode, and offering us a potential example of punctuated equilibrium, we now have Education for Sustainability, led by a gang that says in part that we need to get rid of the old EE, and quickly, because UNESCO tells us we have to "redirect" education towards sustainability. Natural selection, the attractiveness of the programs offered, and government funding, will determine the outcome of this current speciation effort.

We value diversity: this seems to be a core belief for many of us. We are nervous when we see some big players come into our field of education, when we see the big organizations taking all the resources and at the same time presenting a limited range of viewpoints and squeezing out the competition. No monocultures in EE, we might say. Yet not all diversity is sustainable: we have to look for and find the right scale of diversity in EE. We need to especially nourish those organizations and groups that have the strength to survive, those large enough to make an impact, but not so monolithic as to grab all the marbles, and still open enough to incorporate new ideas.

Some other biological metaphors. It looks as if we share, with other biota, the realities of island biogeography theory. For those of you not up on this, the theory, developed in the late 1960s by Robert MacArthur and E.O. Wilson, had a few postulates:

- Diversity increases on new sites by colonization.
- Colonization rate is determined by degree of isolation.
- Extinction rate increases as more species become available;
- Extinction rate is governed by size/complexity.
- Which species go extinct is matter of chance, competition, and environmental change.
- Equilibrium is reached when the 2 rates—Colonization and Extinction—are equal.
- Extinction falls differentially on species: rare species especially are subject to stochastic—or random—effects.

In this scheme, big or complex islands have more species than small or simple ones, islands closer to mainland sources have more species than those far away, and anything that is present in small numbers is always at risk.

So what do we see in EE? We see lots of energy in the larger population centres, with some ability to garner small to medium-level financial resources to create EE products and programs, which are then diffused out into the regions of the province. Further away from the big centres, public schools are smaller and the likelihood is that there are fewer EE teachers. The chance of local extirpation of EE is greater: rare species in small populations are always prone to random effects. When the 1 keen teacher leaves a school, the program he or she sponsors is often gone with them.

So we have to encourage more teachers to be engaged in EE. We have to find ways to increase the size of the EE populations in those small islands around the province, to ensure that the randomness of teaching staff change doesn't have the effect of making EE a locally extirpated species.

Now we know that when we plan biosphere reserves and parks, we try to take into account the understandings we have about island biogeography theory. One prediction made by theory is that large islands are likely to have more species than small ones, and that many small islands, or parks, are likely to lose more species than a few larger islands or parks. Do we, as environmental educators, need to be proliferating organizations and projects in the form of many small "species" of EE, or do we come together, work together, pool our resources, and increase our likelihood of avoiding extinction? We might see a more viable EE culture develop if we reduce somewhat the diversity of programs offered.

Another prediction of theory is the need to connect parks via corridors, especially small parks or parks with very large individuals like grizzly bears, to ensure survival of those species. We need to create corridors and channels for clear movement of individuals and populations. Have we, as environmental educators, put enough effort into making our corridors, our networks, finding programs that can engage lots of teachers and students? I believe that there are fabulous programs around that do this kind of thing: Destination Conservation; the Evergreen Foundation; Linda George's projects with the Wild Bird Trust; Kerrie Post's and Lenny Ross's Colquitz River project; Milt McClaren's Networking BC Rivers. We need to create more corridors, more networks, more connections between us. Let's not keep focusing on the creation of new species: let's look at nurturing those that have the potential to survive and prosper.

We know from more than 20 years of study that the best predictors of a positive environmental orientation to the world are strong childhood experiences of being in natural settings with caring and interested adults. Other than with families, and most of the families in British Columbia now are urban and reasonably detached from wild places, students might have opportunities to have strong experiences with nature by visiting outdoor and environmental schools. These places, like McQueen Lake, North Vancouver Outdoor School, Lake Cowichan Centre, and others, are always running at the edge. They are our endangered habitats, the fragments of what should be a network of environmental schools

Proc. Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15–19 Feb. 1999

affording young people opportunities to live and learn, to study and operate in a way more attuned to natural rhythms and more clearly focused at a sustainable mode of operation. It is not just species that can be endangered, but whole places run the risk of disappearing. Outdoor schools always seem to be close to the cut-off lines that school districts and ministries have for deciding what will be funded and what will not be, and we would all be the less with their loss.

Can we look at EE from an ecosystem perspective, as it does live within an ecology of other institutional frameworks, within ministry requirements, within bureaucratic organizations?

C. S. (Buzz) Holling and associates (Gunderson et al. 1995) have created a way of looking at ecosystem development that has implications for institutional development as well, and should speak to those of us involved in education. But a biblical interlude first. Most of the bible, Jewish version and Christian alike, is a one-way street, a linear progression from Genesis to the end of time. This idea of progress and of temporal directionality has had a profound influence on how we see everything, from human development and the potential to learn, to the evolution of ecosystems from succession through to climax.

Yet we have 1 book, that of Ecclesiastes, that reminds us of the circularity of things, of the recurrence of things, of the creative potential that necessarily fails, only to be reconstructed into new forms—Chapter 3:

1. To every thing there is a season, and a time to every purpose under the heaven:
2. A time to be born, and a time to die; a time to plant, and a time to pluck up that which is planted;
3. A time to kill, and a time to heal; a time to break down, and a time to build up;
4. A time to weep, and a time to laugh; a time to mourn, and a time to dance;
5. A time to cast away stones, and a time to gather stones together; a time to embrace, and a time to refrain from embracing;
6. A time to get, and a time to lose; a time to keep, and a time to cast away;
7. A time to rend, and a time to sew; a time to keep silence, and a time to speak;
8. A time to love, and a time to hate; a time of war, and a time of peace.

I have always liked that book, as it speaks to me of cycles, of growth and change and regrowth and more change.

Holling points out that all systems go through a 4-phase development (Fig. 1), and the rate of movement through the phases varies and the change never ends: there are always "seasons" and times to every purpose. And we might say, especially when we look at the institutional parallels, there are different "heroes" at different stages of development.

The movement from the pioneer phase to a phase of consolidation can be long and slow, and in nature is characterized by increasing strength of connection, and increases in the amount of stored capital—carbon capital, nutrients, or knowledge capital.

In ecosystems, the first stage is characterized by r-selected species: species that can make lots of replicates, species that are opportunists and pioneers, which can begin the process of organizing an environment. Slowly, through time, those species change their environment through the accumulation of biomass, through increasing connections and relationships. Through time, new heroes emerge: species that thrive in stable environments, that produce fewer offspring and are better able to exploit the resources at or near the carrying capacity of their environment. But things happen, and the best-laid natural organizations are laid low, capital is dispersed, connections are broken, things fall apart and chaos follows—from order comes chaos. But now there is lots of good stuff, and life begins again to reorganize—from chaos comes order.

When we manage for endangered species, we have to take into account that nothing in nature is static, that climax is not forever, that randomness and contingency have huge influences.

Institutions and movements can go through the cycle as well, Holling points out. We start, he would say, with the policy phase. Policies are easy to make, many often come out at the same time, they use up a lot of time, and no one has to make much of a commitment to implementation. When I was in Indonesia recently, I was told that simply making policy or passing laws means that the task has been done. No need to do wildlife protection, just pass a policy! Hopefully, some of the policy actually takes hold, is institutionalized, becomes implemented and provides service. These services become expected.

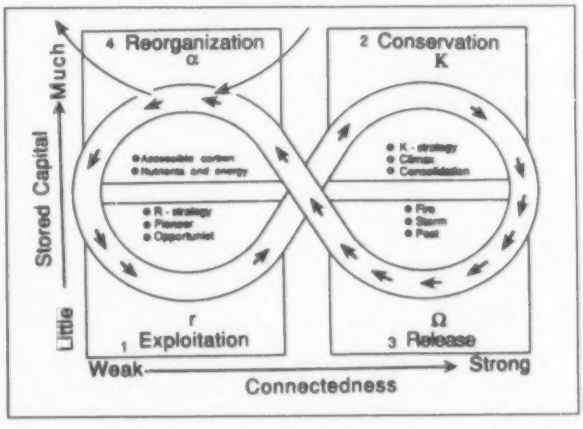

Figure 1. The 4 stages of ecosystem development (from Gunderson et al. 1995).

But the world keeps on keeping on, change occurs around the policies, and the norm is for institutions and bureaucracies to keep doing what they were always doing. Inevitably, there is a crisis—the policy of the past, the programs of the present, are no longer what is needed. Critics, inside and out, point out problems and institutional support may slip away, leading to a state of crisis. At that point, and while things are falling apart, there is room for considering alternatives, reorganizing and restricting our practice until it is again usurped, perhaps, by the decision-makers who bring the whole, now new, program into a process that they can try to control.

Or, as indicated on the top of Figure 1, the whole system can move out into a new sphere of organization, a flip into a new system: a revolutionary evolution.

Where are we in EE in British Columbia? In many ways, we've never seen a fully mature program, never seen a situation where we have really been able to say that EE is an implemented feature of the provincial educational system. And yet, even with our lack of full implementation, we have high expectations and we always feel, in a sense, in a state of crisis. And so we see EEPSA (the Environmental Education Provincial Specialist Association, a British Columbia Teachers Federation subgroup) trying, for example, to create Environmental Educators of British Columbia, trying to show leadership, trying to strengthen connections. Others are doing so as well, and all are needed. Yet what we do always feels unstable, ready to collapse, ready for our carefully sequestered capital to be scattered with no assurance that something or someone will be there to reorganize or, as Holling illustrates, to move us on to the next level. Extinction haunts us all.

OK, enough of the biological metaphors. You get the point—we have to consider some lessons from biology to ensure that we don't make fundamental errors in how we work in this strange field, a field that is always in an underdog position, yet has so much to offer.

There are other reasons, too, why EE might be considered to be on the endangered educational species list. A primary reason might be that many educational practitioners have no idea of the power of EE.

The best place to begin to look at the power of EE might be a recent report published by the United States State Education and Environment Roundtable (Lieberman and Hoody 1998), a study that looked at 40 schools evenly broken down into elementary, middle, and high schools, in 13 states. The authors examined the educational efficacy of teaching and learning within an environment-based educational context, as opposed to a more traditional framework. The findings are remarkable and should act as a strong motivator for those of us working in this area. For example, the observed benefits of using the Environment as an Integrating Concept include:

- better performance on standardized measures of academic achievement in reading, writing, math, science, and social studies;
- reduced discipline and classroom management problems;
- increased engagement and enthusiasm for learning; and
- greater pride and ownership in accomplishments.

If more of our associates were aware of studies like this, I have no doubt we would see more of them take a more serious look at what it is we're promoting and trying to do.

There are other problems. Who teaches EE, and who authorizes its use in schools? Many, if not most of us, are science teachers of 1 sort or another. The folks in the Ministry of Education who create the "curricular space" for us to practice, are likely science teachers as well. Yet for many of us, the approaches of traditional science, the sciences that we were trained in, are not and perhaps never were really satisfying. That may be why many of us are no longer teaching science in schools! Like the student at a secondary school where I presented a workshop on careers in biology, which helped the students look at what science-as-practiced was and whether it was anything like what they did in high school—we are aware that "something is wrong here."

One reason that EE is always in an endangered position is that "something is wrong here." The enablers of EE, the science teachers now working in various ministries of education across Canada, come from a view of science that, for the most part, they learned by rote rather than by having been scientists themselves. Many were students of science and became teachers of science without ever being scientists—but that is another story. They are believers in the traditional view of science: experimental, reductionist, and narrowly disciplinary. Science as we tend to demonstrate it in schools is a "science of parts," a practice based on hypothesis-testing leading to an attempt to "design critical tests for the rejection of invalid hypothesis." The goal is understanding little pieces through the reduction or elimination of uncertainty.

But on campuses across the land, we can see the development of new kinds of sciences: sciences that look at systems rather than at parts; sciences that look more at relationships than to their dissection; sciences where surprise, nonlinearity, and uncertainty are expected, where integration rather than reduction is operational, where historical understanding is vital, and where experiments may be done, but where critical data may come about retrospectively, because it may be that through the after-the-fact viewing of unintentional manipulations of the world we can learn a great deal about how natural systems work.

This new kind of science makes clear that we cannot separate the system under study from those doing the study, because we are in and part of the system. EE, in its incarnation as environmental science study in school, grows out of the former version of science, but will increasingly come to live in the latter. As such, it will make traditional science educators more and more nervous, and that is as it w'll be. Those who

understand and can comfortably live in the different paradigm—and now for the most part it seems to me that they are elementary teachers—will be the early adapters. The rest, hopefully, will catch on. The traditional disciplines in which school science operates do us little good when our global culture is facing such deeply interdisciplinary problems.

Trying to save endangered species by using the traditional approach of isolating individual aspects of problems facing a species, is probably not the way to go. We could, for example, protect lots of habitat for grizzly bears only to find that global warming is changing the plant community composition, or increased rainfall keeps on flooding their favourite foraging area, or that increased UV-b is affecting plant productivity, or that air pollution from a large metropolitan area is resulting in increased sulfur deposition and acid rain, or that ecotourism is driving bears from preferred habitat, or that wholly unexpected and unpredictable events are happening. Indeed, Holling and others now point out that most ecosystem management practices results in more—not less—vulnerable systems, and less—not more—resilient natural systems, along with more entrenched management institutions, and more societies dependent on these increasingly fragile and unpredictable resources.

Maintaining EE in schools is also a complex problem that is going to take some new ways of thinking. We are going to have to find new routes to help our associates embrace interdisciplinary teaching at a time when ministries are focusing more and more towards standardized tests on narrower and narrower themes. My recent project in Indonesia attempted to work with a university science faculty on how they could work together to bring environment- and sustainability-related science education to their students in an integrated way, across traditional subject matter borders. We need to work together to do the same here. We strengthen EE when we bring people together, when we find ways to integrate EE into curricula, when we help students see across boundaries and think in systems terms, when we connect ourselves and our students in a strong and profound way with nat , not worrying about whether it is biology, earth science, or geography.

And why do all of this? There is always value in learning, value in and of itself. But that value becomes diminished when we see the state of the world today, when we hear about what is happening to global systems due to human activity. What is the educational lesson of endangered species for us today?

One danger in teaching about endangered species is thinking that simply telling people about what is happening will be enough to bring them around to what you think should be done. That supposition is not good education, and while it may be what we all wish to see—that is, everyone thinking our way—it is not good education. As Marris (1975: 166) writes:

"When those who have power to manipulate changes act as if they have only to explain, and when their explanations are not at once accepted, shrug off opposition as ignorance or prejudice, they express a profound contempt for the meaning of lives other than their own. For the reformers have already assimilated these changes to their purposes, and worked out a reformulation which makes sense to them, perhaps through months or years of analysis and debate. If they deny others the chance to do the same, they treat them as puppets dangling by the threads of their own conceptions."

Another problem with simply teaching about endangered species without seeing the larger context, is that it is such a downer, and as OISE/University of Toronto professor Edmund Sullivan repeatedly says, "As educators, despair is not our business." We can't move people by bumming them out.

We have to find room within our present curriculum for those species at risk to speak to us; to tell us what the richest, most privileged of the world's citizens are doing to cause their imminent demise. The question of biodiversity reduction is, at its roots, a deeply moral question, a question of moral action. As scientists, and as educators, we are deeply uncomfortable talking about questions of morality, and we have little skill, most of us, in either moral analysis or instruction. Yet what is always put to us is "what do we do?" And at the end of the day, we cannot escape the fact that what we do is grounded in our sense of right and wrong.

If we can hear that the problem of endangered species is part of a larger problem that cannot be shirked by us, and not left to our students, we can work together towards right actions around the larger problem. If we do not see the loss of species as the problem in itself, then we'll be doing something good. When we begin to let endangered species teach us that we are making serious bets on the future, and that our children are going to live in that future and they are going to have to cover our bets, maybe this will be something that gets us thinking and doing. When we look at our children's faces, we begin to know that something had better be starting, and that we're the ones to start it.

I'd like to conclude with a story about a treasure, because deep down, I'm sure, most of us feel that the species we share this planet with are a treasure beyond compare. This story has been told in many ways, but I'm going to present the version adapted by Rabbi Gedaliah Fleer from the stories of Rebbe Nachman of Bratslav. It's called, "The Treasure under the Bridge."

There was once a poor, pious Jew who lived in the city of Prague. One night he dreamt that he should journey to Vienna. There, at the base of a bridge leading to the King's palace, he would find a buried treasure.

Night after night the dream recurred until, leaving his family behind, he traveled to Vienna to claim his fortune. The bridge, however, was heavily guarded. The watchful eyes of the King's soldiers afforded little opportunity to retrieve the treasure. Every day the poor Jew spent hours pacing back

and forth across the bridge waiting for his chance.

After two weeks time one of the guards grabbed him by the lapels of his coat and demanded gruffly, "Jew! What are you plotting? Why do you keep returning to this place day after, day?" Frustrated and anxious, he blurted out the story of his dream. When he finished, the soldier, who had been containing his mirth, broke into uncontrollable laughter.

The poor Jew looked on in astonishment, not knowing what to make of the man's attitude. Finally, the King's guard caught his breath. He stopped laughing long enough to say, "What a foolish Jew you are, believing in dreams. Why, if I let my life be guided by visions, I would be well on my way to the city of Prague. For just last night I dreamt that a poor Jew in that city has, buried in his cellar, a treasure which awaits discovery."

The poor Jew returned home. He dug in his cellar and found the fortune. Upon reflection he thought, the treasure was always in my possession. Yet, I had to travel to Vienna to know of its existence[1].

The earth is our treasure inasmuch as we are one of the earth's treasures. Where we are is always where we can begin. And staying within my Jewish tradition, in the words of the rabbis, "If not now, when?"

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Species Risk Designation at the Canadian Federal Level: A Changing Role for the Committee on the Status of Endangered Wildlife in Canada

David M. Green
Redpath Museum, McGill University
859 Sherbrooke Street West, Montreal, PQ, H3A 2K6, Canada
davidg@shared1.lan.mcgill.ca

ABSTRACT

COSEWIC, the Committee on the Status of Endangered Wildlife in Canada, has for more than 20 years determined the national status of wild species considered to be at risk of extirpation or extinction in Canada. Status determinations are best judgements based upon available scientific information embodied in detailed status reports. They are predictions of a species' probable fate if present trends and conditions continue. The contracting, editing, and preparation of status reports are in the hands of taxonomically-based subcommittees of COSEWIC and their chairs. COSEWIC currently has a permanent membership consisting of the subcommittee chairs plus federal, provincial, and territorial jurisdictions and agencies, and non-governmental organizations, each of which sends delegates. COSEWIC has considered 456 species since 1978; 307 species are now on a respected national "List of Species at Risk," which grows each year. Pending federal legislation will necessitate a number of changes to COSEWIC, including new formal terms of reference, criteria for determining status, credentials for members, and procedures for appointment of members as individuals. Amid much debate, a new and different COSEWIC is emerging, which will report directly to government ministers. Preservation of an impartial scientific process of status designation and the continued independence of the committee are being questioned. An independent, scientific, and apolitical COSEWIC is the cornerstone of a workable and worthwhile Canadian Species at Risk Act.

Key words: Committee on the Status of Endangered Wildlife in Canada, COSEWIC.

The extinction risk for a species (its level of endangerment) is a probability that all the populations of that particular species will go extinct within a certain period of time. The probability, for instance, might be a 10% chance of extinction in 100 years, or 20% in 50 years, or 50% in 10 years, in order of increasing pessimism. In practice, computation of such probabilities is extremely difficult, both because the analyses are complex and fraught with assumptions, and because data are usually limited in scope and accuracy. Consequently, population viability analyses have only been done on a small group of species. Nevertheless, endangered species lists are produced by a number of agencies, using broadly subjective categories of risk (endangered, threatened, vulnerable) rather than probabilities. Status designations are predictions, not descriptions, and constitute judgements on the probable fate of a species if present trends and conditions continue.

Endangered species lists are produced and circulated by 2 different sorts of organizations, largely for 2 different purposes. Conservation groups make lists for publicity and public awareness of endangered species, and to identify causes for concern. Governments make lists also, to demonstrate their action and concern, and to incorporate levels of necessary action and prohibition into laws and regulations. In Canada at the federal level, the agency that determines species extinction risk is COSEWIC, the Committee on the Status of Endangered Wildlife in Canada (or, in French, CSEMDC, le Comité sur le Statut des Espèces Menacées de Disparition au Canada). In its official parlance, COSEWIC determines the national status of wild species, subspecies, varieties, and nationally significant populations of all native fish, amphibians, reptiles, birds, mammals, molluscs, lepidoptera, vascular plants, mosses, and lichens that are considered to be at risk in Canada, based on the best available scientific information. Uniquely, COSEWIC incorporates the goals both of governments and of non-governmental organizations and independent scientists to achieve its ends.

COSEWIC was conceived in May 1976, with the proposal that the Federal-Provincial Wildlife Conference, composed of

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Table 1. Chairpersons of COSEWIC

Years	Name	Affiliation
1977–1987	J. A. (Tony) Keith	Canadian Wildlife Service
1988–1991	W. T. (Bill) Munro	British Columbia Wildlife Branch
1992–1995	Christopher Shank	Northwest Territories Renewable Resources
1996–1997	Dr. Erich Haber	Canadian Museum of Nature
1998–	Dr. David M. Green	McGill University

the wildlife directors, strike a standing committee consisting of federal and provincial government representatives, as well as conservation and scientific organizations, in order to establish the status of endangered and threatened species and habitats in Canada. That proposed standing committee was COSEWIC, which was formed in September 1977 and made its first designations in May of the following year.

Currently, and in effect for its 1999 annual meeting, COSEWIC is a committee of 26 voting members, which are, in effect, not people but jurisdictions, agencies, and organizations, who send representatives, plus the chair (Table 1), who is elected from the membership. These members are COSEWIC's 6 taxonomically-based subcommittees (Birds; Terrestrial Mammals; Fish and Marine Mammals; Reptiles and Amphibians; Vascular Plants, Lichens and Mosses; and Invertebrates [Lepidoptera and Mollusca]), the 10 provinces and 3 territories (including the new territory of Nunavut), 4 federal agencies (Canadian Wildlife Service, Parks Canada, the Canadian Museum of Nature, and the Department of Fisheries and Oceans), and 3 non-governmental organizations (World Wildlife Fund Canada, Canadian Nature Federation, and Canadian Wildlife Federation).

The subcommittees are usually represented by their chairs or, in the case of most of the subcommittees, their co-chairs. The chairs and co-chairs are the most independent of COSEWIC members. They do not necessarily have organizational affiliations, and they include university academics and independent scientists chosen to represent their particular taxonomic groups based upon their expertise and biological knowledge. Scheduled for next year, the Fish and Marine Mammals subcommittee will be divided into 3 new subcommittees: Marine Fish; Freshwater Fish; and Marine Mammals. This will bring the number of subcommittees (to be redubbed Species Specialist Groups) to 8, and thus bring the voting membership of COSEWIC to 28, plus the chair.

Aside from the voting membership representatives, COSEWIC also consists of: the taxonomic subcommittees themselves; a Standing Committee composed of the subcommittee chairs, which coordinates activities of the subcommittees; and a Secretariat, which is staffed by the Canadian Wildlife Service and looks after operations.

The assessment process followed by CCSEWIC begins with the subcommittee chairs/co-chairs, whose responsibilities include: identifying species to investigate; finding authors to write species status reports; preparing the contracts for the reports and forwarding them to the Secretariat; editing the reports when they come in; and presenting the reports and status recommendations to the COSEWIC committee. The subcommittee chairs/co-chairs thus shoulder most of the responsibility for making COSEWIC work. The Species Status Reports that are commissioned by the subcommittee chairs contain, as much as possible, best available information on species, including range, habitat, general biology, protection, special significance, changes and trends in population size and distribution, limiting factors, and actual and potential threats. Each report, written by a knowledgeable expert on the species, also includes an evaluation and proposed status. Once received, a draft report is reviewed by appropriate subcommittee and range jurisdiction(s), and by members of COSEWIC. If a species has been previously reviewed, a Status Report Update must be commissioned, ideally every 10 years, or when information indicates that its status may have changed. An update may conclude 1 of several possible outcomes: status reconfirmed; an uplisting into a more serious category of risk; a downlisting into a less serious category of risk; or a delisting, which removes the species from the List of Species at Risk.

In the assessment process, the taxonomic subcommittees are there to provide scientific expertise on taxonomic groups and review draft reports for scientific accuracy and completeness, rather like an editorial board. The subcommittee may recommend acceptance or rejection of the report and also recommend a status designation. Taxonomic Subcommittee members perform other functions, as well, in helping to develop candidate species lists, commission status reports, and in accepting reports from interested third parties.

Reports that have been approved by the subcommittees and vetted by the relevant jurisdictions are brought by the subcommittee chairs/co-chairs to COSEWIC's annual general meeting, held every April. This is a closed meeting of members and observers held to evaluate species of concern. Precise and detailed information is often presented, and so, to protect the species, that information is not made public. Instead, a public forum is held to announce the results after the decisions are made. During the meeting, for each species, the relevant subcommittee chair presents the species status report and a status designation is put to a motion. The motion may be debated, but decisions are reached, as much as possible, by

Table 2. COSEWIC endangerment risk categories. Endangered, Threatened, and Vulnerable comprise the "At Risk" list.

Category	Definition
Endangered (E)	species faces imminent extirpation or extinction
Threatened (T)	species is likely to become endangered if limiting factors are not reversed
Vulnerable (V)	species is sensitive to human activities or natural events
Not at Risk (NAR)	species is secure
Extirpated (XT)	species no longer exists in the wild in Canada
Extinct (X)	species no longer exists
Indeterminate (I)	insufficient scientific information on which to judge

consensus. If the committee's opinion is not initially unanimous a vote may be taken; a two-thirds majority is required for acceptance of a motion. If the proposed status is not accepted, a motion for a different status is proposed, debated, and voted upon. Controversies usually arise because risk levels are somewhere along a continuum of probabilities, but designations are discrete categories. It often requires discussion to determine on which side of the line between categories a species' risk level may actually lie. There may still not be unanimity, but a two-thirds majority win also means that no member of the committee can have a veto over designations.

The National List of Species at Risk in Canada, produced by COSEWIC, now has 307 species on it, out of the 456 species COSEWIC has considered since 1978. Ten species are considered Extinct, 12 Extirpated, 73 Endangered, 72 Threatened, and 140 Vulnerable; a further 121 are currently considered Not at Risk, and 19 are considered Indeterminate (see definitions in Table 2). Out of the 67 species re-examined with status report updates, 9 have been delisted. This year (1999) some 80 reports, about half of which are updates, will be considered by COSEWIC and the list will grow. This, of course, does not reflect the speed at which species are becoming at risk. It reflects the pace at which COSEWIC can investigate them. Hundreds of candidate species remain to be assessed and each species takes time to assess, report on, and review. Mainly because of updates, the number of reports considered by COSEWIC grows each year, and that is expected to continue.

The result of COSEWIC's work, the National List, is understood and respected by all jurisdictions and conservation groups. There are good reasons for this influence. It is scientifically based and produced in a climate relatively free from political shenanigans. All members' views are aired, and it remains independent, at arm's length from governments. Its particular strength is its inclusiveness, incorporating government, non-government, and independent scientists, such that no single group can unduly influence the outcome.

Thus, COSEWIC works. It works because of its broad range of expertise with wildlife. It works because it is cost effective, being the result of mostly volunteer effort. It works because it is respected for integrity, impartiality, and independence. It works because it is inclusive and enjoys "buy-in" from governments, non-government organizations, and the scientific community. It works because of its continuity and effective consensus building. It also has a public education role and makes its status reports freely available once approved.

There is room for improvement, though, and there are limits on what COSEWIC can do. It cannot be held responsible for provincial designations, and not all wildlife in Canada is represented on its subcommittees. COSEWIC does not currently have a subcommittee for crustaceans, for example. COSEWIC could be castigated for being ineffective in publicizing and promoting itself in the past. And COSEWIC's increasing workload will require solutions.

One improvement currently being debated by COSEWIC is adoption of a rigorous system of standardized criteria. Criteria for designations need to be defensible, and they are helpful if they can act as consistent guidelines for evaluating the evidence contained in status reports. This will aid in yielding consistent and readily understood justifications for categorization of extinction risk. To that end, the IUCN (International Union for the Conservation of Nature) evaluation system was adopted 1998, but the COSEWIC categories, rather than the IUCN categories of risk, were retained. COSEWIC's designations and the categories of Endangered, Threatened, and Vulnerable, as COSEWIC uses them, are well understood in Canada and are used in current and pending regulations and legislation. To abandon them would be a mistake. So, it remains to reconcile the IUCN method with the COSEWIC nomenclature (Fig. 1).

Impending federal endangered species legislation will mean increased importance for COSEWIC. The original Canadian Endangered Species Protection Act (CESPA, or Bill C-65) died on the order paper in 1997, and new legislation—a "Species at Risk Act"—is to be introduced this spring in harmony with a National Accord reached with the provinces and territories. COSEWIC henceforth will report to a newly established Canadian Endangered Species Co-ordinating Council (CESCC), composed of ministers of wildlife from the provinces and territories and the federal government. This will raise the stakes for COSEWIC, as all action on endangered species at the federal level will start with COSEWIC's designations. COSEWIC will report directly to the ministers and its designations will (eventually) carry legal force.

Tinkering with COSEWIC has already begun. The 6

subcommittees will become 8 Species Specialist Groups. Appointment of members has not yet been determined; many models are under consideration. The non-governmental representatives have not yet been clarified. There has been pressure from aboriginal groups to include traditional ecological knowledge (ATEK) at all levels, and assorted stakeholders' views argue for a stronger or a weaker COSEWIC. COSEWIC designations will have legislative implications, so it is logical to expect jurisdictions to aim to influence COSEWIC's behaviour. Tensions arise because, fundamentally, conservation organizations and governments do not have the same goals in creating endangered species lists.

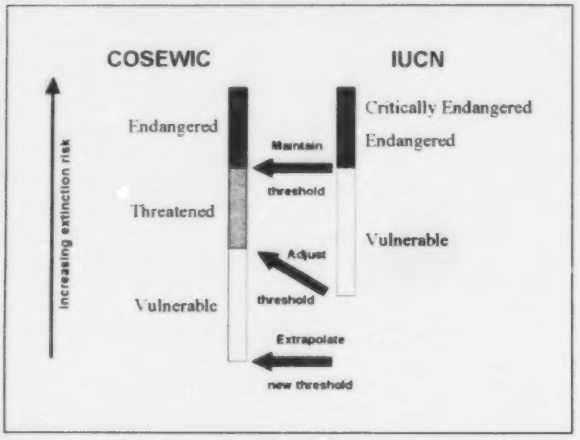

Figure 1. Alignment of IUCN (International Union for the Conservation of Nature) and COSEWIC risk categories showing the adjustments to criteria threshold values that may reconcile the 2 systems for COSEWIC's use.

Some points, however, are clear. Politics cannot be avoided, because designating status is itself a political act, aiming to influence human awareness and behaviour. But COSEWIC has no legislative powers, nor can it, because legality is the constitutionally mandated responsibility of governments. This is not a weakness for COSEWIC because, freed from legislative onus, COSEWIC can consider species solely on the facts and keep the politics out of the process. The ability to publish its list gives COSEWIC its clout.

Can COSEWIC continue to be independent and scientific under legislation? Despite competing pressures, it has to be, for if not, it will become irrelevant. The determination of species status must not be done under external political pressure or political control; the facts alone should rule in COSEWIC. An independent, scientific, and apolitical COSEWIC is the cornerstone of a workable and worthwhile Canadian Species at Risk Protection Act.

Improving the National Recovery Program in Canada

Cathy Merriman
Endangered Species Program, World Wildlife Fund Canada
245 Eglinton Avenue East, Suite 410, Toronto, ON, M4P 3J1, Canada
cmerriman@wwfcanada.org

Kent Prior
Endangered Species Division, Canadian Wildlife Service
Hull, PQ, K1A 0H3, Canada

ABSTRACT

Recovery planning at the national level for species at risk in Canada has been coordinated by RENEW (Recovery of Nationally Endangered Wildlife) since 1988. To date, most of this work has focused on terrestrial vertebrates, and a total of 19 recovery plans have been approved. Implementation of those plans has been supported by government agencies and others, totalling $27 million over 10 years. One of the main national programs to support implementation of recovery plans has been the Endangered Species Recovery Fund (ESRF), a partnership between World Wildlife Fund-Canada (WWF-Canada) and the Canadian Wildlife Service (CWS). As the list of species at risk in Canada continues to grow, and as the time approaches when we will have federal species protection legislation in place, we need to increase and improve our capacity to produce and implement recovery planning documents. To that end, a national Recovery Working Group (RWG) has been formed and is developing a series of recommendations that will form the basis of a new national recovery program. At the same time, WWF is supporting the development of new, needed recovery plans that can test the proposed streamlined models for these plans.

Keywords: COSEWIC, private sector, recovery planning, RENEW.

In 1988, Canada's wildlife ministers established a committee known as RENEW, or "Recovery of Nationally Endangered Wildlife," mandating the group to coordinate the recovery of species at risk at the national level. The primary original goal of RENEW was to coordinate recovery teams, who would produce plans that outline recovery actions for species listed by COSEWIC (the Committee on the Status of Endangered Wildlife in Canada). The general goal of those plans is to improve the conservation status of the species in question. Working to increase public awareness of species conservation issues in Canada has also been an important goal of RENEW.

While RENEW's mandate covers all species listed by COSEWIC, the practical focus to date has been mainly on terrestrial vertebrates. Most of the plans produced so far concern a single species, but the first multi-species and ecosystem plans have now been initiated. For example, teams are working on a plan for the hooded warbler (*Wilsonia citrina*) and the Acadian flycatcher (*Empidonox virescens*) in Ontario, and another for the South Okanagan Valley ecosystem.

Since its formation, RENEW has approved 19 recovery plans. Teams are currently in place for another 33 species. The current COSEWIC list, however, contains approximately 100 species and populations with endangered or threatened status for which plans still need to be initiated. This is a concern to many in the Canadian conservation community because the COSEWIC list will continue to grow. It is likely that federal species protection legislation to be introduced later this year will include a requirement that recovery plans be developed for endangered and threatened species within relatively short time frames. Thus, the Canadian capacity to produce such plans must be increased and improved.

RENEW recovery teams comprise government and non-government partners, in particular, scientific experts and relevant jurisdictions for the species in question. Some teams also have private-sector stakeholders as members. This type of membership seems likely to increase in the future for a number of reasons, perhaps most importantly because many types of recovery work will only succeed with the wholehearted acceptance and participation of people from many sectors of society.

Various agencies and organizations have spent a total of about $27 million on the preparation and implementation of recovery plans. Recovery plans are intended to function as practical guides to activities that are needed to help species recover, thus it is important that government agencies and others make the commitment to implement the plans once they have been approved by RENEW. In many cases recovery work is already underway by the time a plan is completed and approved. While nationally-coordinated recovery planning and implementation have been going on for more than 10 years in Canada, actual successes are difficult to measure. This is at least in part because species recovery itself is a

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb. 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

long-term process, often requiring substantial resource commitments and long timelines. Perhaps the best example of a "success story" is the peregrine falcon. The national recovery plan for the *anatum* subspecies has guided activities carried out by many partners, and, after 2 decades, the peregrine is making a comeback in many regions of Canada.

One of the largest national-level programs to support the implementation of recovery plans has been the Endangered Species Recovery Fund (ESRF). The ESRF has existed as a partnership between World Wildlife Fund-Canada (WWF-Canada) and the Canadian Wildlife Service (CWS) since 1988. One of the strengths of the ESRF is its requirement that grant recipients bring at least 50% of their project budget as matching funds, which means that the approximately $3.5 million in grants disbursed over the years has contributed to at least $7 million in conservation activities. This matching requirement is intended to foster broad partnerships in the conservation of species at risk and their habitats. The majority of ESRF support has been for COSEWIC-listed species, especially endangered and threatened ones, but work on species in other categories may be supported as well. Another strength of the ESRF is the independent scientific review that all proposals undergo. Projects supported include elements of research, direct recovery actions, and public outreach. In 1998 and 1999 the number of requests for support from the ESRF increased dramatically compared to all previous years, and we expect that this trend will continue.

In October 1996, Canada's provinces and territories agreed in principle to an "Accord for the Protection of Species at Risk in Canada." The accord encompasses commitment to measures such as complementary legislation and programs to provide legal protection to species at risk and their habitats. It also highlights the need for the development of preventative measures, including stewardship programs and recovery plans, and for an independent assessment process for the listing of species. The accord was developed partly in response to planned federal species protection legislation, and to address the recovery needs of the growing number of species on the COSEWIC list. Cooperation and harmonization among the jurisdictions will be fundamental to the success of ongoing and future recovery work. Since the new federal legislation is expected to require the development of recovery plans, a system to ensure that recovery plans are comparable across jurisdictions will be an important element of Canadian conservation programs. The accord also states that the signatories will seek guidance for implementation of the accord through an evolving national framework for species at risk.

That guidance was first sought at 2 meetings of national recovery team chairs in early 1998. At these brainstorming sessions, one in Regina and one in Toronto, recovery team chairs raised many issues pertinent to improving the recovery planning and implementation system in Canada. Among those issues was a consensus that the species recovery system needs to be more efficient, especially during the plan approval phase. It was agreed that the emphasis needs to be on recovery actions rather than the formal planning phase. Recovery team composition was discussed, especially regarding the future involvement of private-sector stakeholders on teams. In the past this has not happened on most teams. Some feel that this will be necessary to ensure general private sector acceptance of proposed recovery activities. Others feel that this will compromise the conservation integrity of plans, and that stakeholders should only be included at the implementation stage. There was a general consensus among team chairs that in future we need to focus more on coarse-scale conservation, taking ecosystem and multiple-species approaches. Perhaps the greatest consensus was that there is a shortage of resources, both financial and human, in Canada to adequately address the recovery needs of species at risk.

Later in 1998 the wildlife directors struck the Recovery Working Group (RWG), requesting that this group explore options and make recommendations for improving the Canadian recovery system. The RWG is composed of federal jurisdictions (CWS; Department of Fisheries and Oceans), provincial jurisdictions, and conservation non-government organizations (WWF; Wildlife Habitat Canada). The group will make specific, practical recommendations to the wildlife directors, which they will then use to form the basis of a new National Recovery Program

As a next step, Environment Canada hosted 2 stakeholder workshops on federal species protection legislation and the accord. The RWG held sessions focusing on the role of stakeholders in recovery planning and activities. Among others, representatives of agricultural and livestock associations attended, as did aboriginal groups, resource users such as fishermen, and other industries. Those attending the meetings expressed general acceptance of the principles that species recovery needs to be based first on good science and on meeting the ecological requirements of species, then take into account socio-economic factors. At the same time, many important questions were discussed and not resolved during these meetings. These included concerns about incorporating traditional ecological knowledge (TEK) into recovery planning, stakeholder participation in recovery teams, and methods for considering socio-economic needs relative to recovery planning.

The information generated at these different meetings gave the Recovery Working Group a starting point to devise recommendations for the operation of an improved recovery system. One of the main goals of the RWG is to develop a template for a streamlined recovery plan model that will facilitate the timely production of documents that will, in turn, guide recovery activities for species that become listed by COSEWIC. To do this the group must take into account the strengths of the existing RENEW model and combine

them with the means to address the restraints that will be imposed on the process by time and financial limitations. The group has formed 4 working groups, addressing the following principal issues:

- Priorities, among species at risk needing plans, and among actions within plans.
- Planning and implementation, including the function and structure of recovery teams. This group will also attempt to develop tools and provide guidance for the production of recovery plans and implementation schedules.
- Review, approval, and progress. This group will attempt to devise a streamlined procedure for approving recovery plans and schedules. They will also develop mechanisms for evaluating recovery success and reporting to appropriate agencies.
- Funding requirements and sources. This may be one of the most important issues to address, because long-term commitment of resources will be essential to the success of any new recovery program in Canada.

As of this writing (March 1999), the Recovery Working Group is still working on all of these elements of a new recovery program, and is open to input from all interested parties with experience in or concerns about these areas.

While this work is continuing, WWF has in the meantime developed an opportunity to put the new, evolving model into practice. Through a contribution from the federal Millennium Bureau, WWF is offering financial support for the development of recovery plans for species currently listed but lacking recovery plans. In this way we hope to test some of the new models being proposed and to help clear up some of this "backlog" of listed species in need of more coordinated recovery work.

Genes, Species, Ecosystems: Is Surrogacy Meaningful?

Jim Pojar
British Columbia Ministry of Forests
Postal Bag 5000, Smithers, BC, V0J 2N0, Canada
Jim.Pojar@gems3.gov.bc.ca

ABSTRACT

Biological diversity encompasses genes, species, and ecosystems. Some suggest dispensing with the habitat or ecosystem level of biodiversity, for a variety of reasons—which I will explore. They choose species diversity as a surrogate for biodiversity. The focus on species, indeed on only certain groups of species, is ill-advised, and will inevitably result in negative consequences for some taxonomic groups. Making biological diversity synonymous with species diversity trivializes the broader meaning of biodiversity and promotes misconceptions of conservation issues. Thorough sampling, inventory, and monitoring of species, populations, and gene frequencies require far too much time and effort to underpin a practical conservation strategy. Management for biodiversity must concentrate on the ecosystem and landscape levels of organization; in other words, must employ a "coarse-filter" strategy, with tactical application of the "fine filter."

Keywords: biological diversity, coarse filter, communities, ecosystems, fine filter, species diversity.

This is a talk I should not have to give. I will present little that most of you have not read or heard before. Much of it falls in the category of "what should be self-evident, at least to biologists." But judging from some recent public and academic discourse, and recent trends in resource management in British Columbia, this material bears repeating. Please note that many of my points are in the context of terrestrial environments, in particular of forests and forestry.

COMMUNITIES AND ECOSYSTEMS: REAL OR ILLUSORY?

Some biologists maintain that we should not use communities or ecosystems as measurable, operational targets in managing for biodiversity because: a) communities/ecosystems are conceptual, not very concrete units; and b) they are naturally unstable, continually moving targets (Bunnell et al. 1998, Simberloff 1998*a*). Bunnell et al. (1998) pointedly exclude habitat or ecosystem diversity as a component of biodiversity, which becomes the "diversity of genes, populations, and species."

Communities and ecosystems are concepts, but so too are the gene and the species. In particular, there continues to be much discussion in biology about what exactly a species is (Mallet 1995). Certainly communities and ecosystems are less discrete (I would not say they are less concrete—these assemblages do exist) and repeatably identifiable than are genes or species, which is what one would expect for a higher level of generalization, a higher level of biological organization. One can argue that the generality of the ecosystem concept is one of its chief strengths (Burns 1992, Kay and Schneider 1994). Note also that the ecosystem is still considered the most influential idea in ecology, judging by recent textbooks of ecology and surveys of professional opinion, such as Cherrett (1989).

Most certainly ecosystems change; no one could legitimately argue otherwise. But change includes rate and extent, and depends on scale, both in time and space. Individual forest ecosystems (or developmental stages of a given forest stand) continually change, but some change faster than others. In some old forests, especially in humid forests dominated by gap dynamics, the system can be characterized as a "shifting mosaic–steady state" (after Bormann and Likens 1979), with little evidence of change at the landscape scale for hundreds and even thousands of years (Lertzman et al. 1996). Lertzman and colleagues now have data from Clayoquot Sound (Gavin et al. 1997) and elsewhere demonstrating that some such forests have gone for several thousand years between stand-destroying disturbances. Paleobotanists tell us that extant, wet coastal forests have not changed all that much in the last 3–4,000 years, although in the past 10–12,000 years they have changed considerably (Hebda and Whitlock 1997).

But there are deeper undercurrents here. The objections (especially from within the cloister of academic ecologists) that communities and ecosystems are not real, reflect the persistent debate about the individualistic vs. the typic concept

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of ecological communities. Barbour (1996) provides a revealing analysis of this debate in a case study of a revolutionary shift in the plant community paradigm during the 1950s, a shift that suggests that culture can lead science, rather than the other way around. The shift was from the Clementsian organismic concept to the Gleasonian individualistic concept of the plant association, from plant communities as discrete units (organic entities) to the continuum theory of vegetation. From a broader perspective, what happened represents yet another example of holism yielding to reductionism, a common theme in the history of science.

The cultural interpretation: post–World War II U.S.A. was disaffected with socialism; holism in ecology and culture became unfashionable; individualism and existentialism rose to prominence (individual self-absorption over social involvement), as did the themes of antiestablishmentarianism (beatniks) and social fragmentation—themes that still resonate in contemporary American society. "It could be that ecologists consciously or subconsciously swam with the currents of their surrounding culture to change some fundamental opinions, to take new research directions, and to view nature in a reductionist, rather than a holistic way" (Barbour 1996).

In North America, the debate has been largely one-sided (the Gleason, individualistic side) for the past 40 or 50 years. Interestingly, some North American ecologists are beginning to think what was unthinkable—or at least unpublishable—until recently. Namely, that there is more to the organization of communities than mere chance, interference, and species individualism (Callaway 1997). In addition to the manifold organizing impacts of parasites and pathogens (Dobson and Hudson 1986), there is increasing empirical evidence of positive interactions (facilitation) among the primary producers themselves, as well as mutualisms between trophic levels (for example, mycorrhizae, nitrogen fixation, pollination, fungi, and fungivores): see Boucher et al. (1982), Bronstein (1994), Callaway (1995), Kareiva and Bertness (1997), Wilkinson (1998). One can discern the emergence of some qualified support for the concept of communities as organic entities, or at least for the notion that communities represent a combined and structured expression of positive and negative interactions among their constituent species.

So the debate continues (Palmer and White 1994) after all these years—Clements and Gleason worked back around the 1920s. Keddy (1993) maintains that the debate over existence of communities is an ontological and epistemological game that is peripheral and perhaps harmful to scientific progress. After becoming involved in this issue in the late 1960s and early 1970s, and after doing considerable fieldwork, I concluded that it was trivial. Trivial in the sense that vegetation and communities and ecosystems have elements of both discreteness and continuity; often boundaries are indistinct but sometimes they are very sharp, depending on the underlying physical gradients; dominant trees behave differently than understory vegetation; discreteness/continuity vary among different regions and vegetation types; and so forth. O'Neill (1996) expresses this duality very cogently: "The dichotomy between the thermodynamic, biogeochemical paradigm of ecosystems and the evolutionary, competitive paradigm of species is the most basic, though seldom discussed, problem in ecology. The situation is fully analogous to the old wave and particle dichotomy in physics." So it is, but why are ecologists still debating this?

Well, partly because they can still make considerable academic hay by attacking or defending one or the other of these views. As Barbour points out, another reason is cultural; it is a bit like asking why the National Rifle Association is still so powerful in the United States. Furthermore, there are nontrivial consequences for measuring/monitoring/conserving biodiversity if you believe that communities and ecosystems do not exist, or, if they do, cannot be delineated. Simberloff (1998a) suggests that if one is primarily concerned with the services provided by ecosystems and communities, and particularly if one holds the individualistic view, a large degree of species loss and of substitution of 1 species for another could be perfectly acceptable. Some foresters and range managers readily warm to this world view.

Breathing life into what should be a moribund issue also provides a venue for revisionist ideologues. The mechanistic, constant change, chaos-plus-contingency approach to ecosystem organization and function underlies the revisionist view of Nature, conservation, protected areas, landscape corridors, and so forth. This outlook has intellectual ties to "post-modern deconstruction" theory of the humanities (Soulé and Lease 1995), which asserts (among other things) that definitions of Nature are culturally based, that what biologists perceive as the reality of Nature is actually a construct of First World, largely white, male scientists, and has no more validity than competing constructs. Competing constructs could be: Nature as what you struggle with to survive; Nature as capital to be liquidated, or wilderness to be industrialized; Nature as a shamanistic vision; Nature as a Disneyesque theme park or a plastic diorama. Nature does not really exist—it is a subjective, culturally constructed concept, an artifact of human desires. Many so-called natural areas were in fact modified and managed by indigenous peoples prior to European settlement; every corner of the globe today has been affected by human activity; everything changes constantly; the "natural" state cannot be objectively defined. Therefore, the deconstructionists argue, it is meaningless to speak of preserving or restoring Nature.

O'Neill (1996) clearly proclaims a somewhat related, mechanistic view of the ecosystem: "After all, the ecosystem is merely the localized, transient experiment in species interaction. It is mostly designed to support life under the temporary local conditions. When it collapses, due to disturbance

or changing conditions, no great tragedy. The species, the enduring, stable entities, simply reassemble a new, more appropriate ecosystem."

Some well-known ecologists, such as Botkin and Simberloff, seem to subscribe to this view, or at least some of their work has been interpreted to support it, often by interests that are advanced by such an interpretation. Some other, equally well-known ecologists have a very different view of the value of natural ecosystems and the tragedy of their demise. More importantly, such views have as much to do with the individual ecologist's philosophy of science and cultural conditioning, as they do with objective truth or scientific credibility.

This could appear like a lot of fuss over a simple decision. But it is important to appreciate the intellectual and cultural antecedents of certain "schools of thought" (perhaps better termed "systems of belief"). Regardless of their belief system, biologists should not dismiss the coarse-filter, ecosystem-based approach to conservation of biodiversity so briefly and conveniently, or at least should be very clear about the reasons for dismissal.

One must also confront management issues. In practice, it is very difficult to manage a gradient; regulatory structures and operating procedures virtually impose compartmentalization. But even ecologists who (like me) believe that "most communities intergrade continuously along environmental gradients, rather than forming distinct, clearly separated zones" (Whittaker 1975) can, if pressed, describe, classify, and map communities, ecosystems, and landscapes. As long as they acknowledge that any classification (including that of species) is an abstraction, a human construct devised for convenience and communication, no harm done and usually something useful accomplished.

GENES, SPECIES, OR ECOSYSTEMS?

I agree with Bunnell et al. (1998) that "genetic variation cannot be a measurable, operational target" in forest management. But we must not forget that the diversity found within species is the ultimate source of biodiversity at higher levels, and that understanding forest biology and ecology requires at least a rudimentary understanding of the system of genetic variability of forest species. Describing the genetic architecture and monitoring the genetic diversity of certain species of management concern, like commercial tree species, Vancouver Island marmot, devil's club, and pine mushroom, certainly should be part of the stewardship of forest land.

Unfortunately, many people think that biodiversity is synonymous with species diversity, or is sort of a new-age version of species diversity. Even some biologists still conflate species diversity with biological diversity. Hence, we encounter a continuing preoccupation with species, and with ways to quantify their diversity. This preoccupation spawned an academic industry in the 1960s and 1970s (e.g., Whittaker 1972), an industry that shows few signs of abating (e.g., Magurran 1988, Ricklefs and Schluter 1993, Huston 1994, Brouard and John 1997, among many others).

Species-based approaches still have an important role to play in biological conservation (Simberloff 1998*b*), and we certainly need to know more about genetic diversity and about the genetic variation of feature species. But I agree with Franklin (1993) that "larger-scale approaches—at the levels of ecosystems and landscapes—are the only way to conserve the overwhelming mass—the millions of species—of existing biodiversity." Franklin specifically had in mind the "physically smaller but overwhelmingly more numerous elements of diversity" (invertebrates, fungi, bacteria) that perform critical ecosystem functions, such as decomposition, nitrogen fixation, and nutrient cycling—"the little things that run the world."

Some ecologists contend that a strategy for conserving biodiversity would best be organized around the central concept of ecology: the ecosystem (Franklin 1993, Noss 1996). The ecosystem concept unites the abiotic and biotic realms, includes processes as well as entities, and is fuzzy, elastic, and inclusive enough to be applied at different scales and to very different habitats (Noss 1996). Furthermore, most terrestrial ecosystems can be characterized, classified, and mapped—which means that they and the structural elements they contain can be "measurable, operational" targets.

SCALE

Suppose for the sake of argument we decided that species were the "enduring, stable entities," that they and their interactions were paramount, and, therefore, that managing for sustained biological diversity should focus on the species level of organization. We would then confront the problem of scale. Scale in space, time, and organization is a fundamental issue in ecology (Levin 1992).

Consider the concept of levels of biological organization (after Rowe 1961). Biological diversity is more comprehensive than species diversity, so we must specify the biological hierarchy (genetic, taxonomic, or ecological) and the organizational level under consideration. Biologists could try to inventory all the genetic elements, all the taxonomic elements, all the ecological elements of a particular area. Even if we knew all the elements present, "no accepted calculus permits integration of counts of elements across levels within a hierarchy (e.g., phyla and species) or across hierarchies (e.g., species and genes). Arguably, no such calculus should be sought" (Angermeier and Karr 1994). We must seek simpler assessments, but I maintain that a focus on the species level of the taxonomic hierarchy is misplaced.

COGNITIVE DISSONANCE

Failure to conceptually integrate the multiple aspects of biodiversity results in narrowly conceived and incomplete comparisons (Angermeier and Karr 1994), like between-habitat species richness of vertebrates, or indices of taxonomic diversity based on cladistics and distinctness of taxa (Humphries et al. 1995). Such analyses are interesting and valuable, but not comprehensive or even representative assessments of biodiversity—because they ignore genetic and ecological hierarchies. Making biological diversity synonymous with species diversity trivializes the broader meaning of biodiversity, and promotes misconceptions of conservation issues. Loss of biodiversity is much more than species extinctions, which often represent the endpoint of a process of population extinctions resulting from habitat destruction and modification. In many cases, species do not go extinct but are lost from much of their former range, (e.g., grizzly bears). Conversion of huge areas of old-growth forest to short-rotation managed forest, extensive overgrazing of grasslands, declines in hundreds of genetically distinct salmonid stocks, and the loss of distinct populations from portions of a species' range, all represent significant losses of biodiversity, regardless of whether any species become extinct. Addition of species of invasive aliens, often with cascading negative effects on native biota, is another, often-neglected issue (Hobbs and Mooney 1998).

Furthermore, one could conclude from a hierarchical analysis of scale (e.g., Jensen et al. 1996) that for most metabiota (multicellular macroscopic life) except probably arthropods, sampling, inventory, and monitoring at any scale less than 10s or 100s of thousands of hectares, and for less than 10s of thousands of years, would involve assessing changes in populations (dynamics, demographics) or in gene frequencies—not in speciation or extinction, for the most part. One can think of specific exceptions to this generalization, like the Dawson caribou or Vancouver Island marmot or sticklebacks or narrowly endemic plants. Studies of the dynamics of populations and gene frequencies are valuable, but they give us only part of the picture. To provide more than a sketch of what is happening to species (our "enduring, stable entities") would require enormous effort over daunting time, even for a handful of "indicator" species (Niemi et al. 1997). Meanwhile, the ecological face of British Columbia is changing literally before our eyes. Logging proceeds apace (about 200,000 ha per year over the last decade), to say nothing of landscape alteration by agriculture, hydroelectric developments, and urbanization. I am not reassured that logging has apparently not yet caused the extinction of any vertebrate species in British Columbia. We haven't been at it very long (2 rotations at most). The Swedes have been, and almost 2,000 organisms—mostly non-vertebrates—are currently Red-listed (Östlund et al. 1997). In this respect, vertebrate species are decidedly not a good surrogate for "species" in general; if logging causes species extinctions in British Columbia, they are unlikely to occur among the vertebrates (e.g., Winchester and Ring 1996).

VERTEBRATE SURROGATES?

Vertebrates are indeed useful (though partial) indicators of species diversity. Vertebrate species diversity could be an acceptable surrogate for biological diversity if:

1. species are an acceptable surrogate for biological diversity;
2. vertebrate species and their biology and habitat relationships represent well other taxonomic groups and their ecologies; and
3. a focus on vertebrate species does not result in negative consequences for other taxonomic groups.

None of these conditions holds. Forests are not merely woody substrates or growth media for vertebrates. Forests are assemblages of organisms (communities) dominated aboveground by plants (the primary producers) and arthropods, and belowground by soil fauna, microbes, and fungi. Taxonomically and in terms of biomass, vertebrates constitute much less than 1% of British Columbia's living organisms. With respect to biomass and energetics, vertebrates are largely incidental in forest ecosystems—although they (e.g., keystone herbivores) can play disproportionately large roles in ecosystem organization and dynamics.

I strongly disagree that "the sustained presence of terrestrial vertebrates is a credible indicator of the state of other components of biological diversity" (Bunnell et al. 1998). I certainly would not rely on amphibians, shrews, and rodents as indicators of the state of the soil fauna, which is megadiverse in temperate forests. Nor are rodents reliable indicators of fungi. The sustained presence of terrestrial vertebrates cannot tell us much about the state of plant diversity. However, I am not too worried about most species of vascular plants with respect to forest management (but see Jules 1998), or at least am much less concerned about species than about certain forested and nonforested plant communities (Pojar 1980).

Cryptogamic plants are another matter. In particular, forest management threatens some epixylic species that rely on decaying wood for substrate (Berg et al. 1994), and threatens certain epiphytes—especially late-successional lichens (Lesica et al. 1991; Rose 1992; Goward 1993, 1994; Esseen et al. 1996; Sillett and Neitlich 1996). Goward and Arsenault have more to say about lichens and forests elsewhere in these proceedings. I merely emphasize that maintaining the epiphytic lichen component of biodiversity requires maintaining forests in all age classes, including very old stands. Widespread short-rotation forestry reduces such diversity (Fig. 1). To assess the state of epiphytic lichens in forests, it makes more sense to monitor the lichens themselves, or the

status of the old forests in which the threatened lichens thrive, than to monitor caribou and flying squirrels.

Lichens are peculiar organisms, but it is unlikely that epiphytic lichens are unique in their relationships with humid forests. I suspect that epiphytes in general behave in a somewhat similar fashion, and perhaps also groups like canopy arthropods. Lowman and Nadkarni (1995), Pettersson et al. (1995), Shaw et al. (1996), and Humble et al. (2000) address these issues.

One could argue that lichens and other epiphytes are not that important in the scheme of things, or that there is a lot of redundancy in the epiphytic guild. But the sheer biomass of epiphytes in some old forests, and the role of epiphytes in nutrient cycling and as habitat for other creatures, suggest ecological significance (see also Rhoades 1995). Moreover, functional redundancy does not necessarily imply ecological dispensability (Naeem et al. 1998, especially their Fig. 1).

CONCLUSIONS

In summary and in my opinion:

- Species diversity is not an acceptable surrogate for biological diversity.
- Vertebrates are not acceptable indicators of species diversity of all biota.
- More attention must be paid to loss of populations (and genotypes) through ecosystem modification and habitat loss, and to gain of species via invasions.
- Management for biodiversity must concentrate on ecosystem and landscape levels of organization.

This is the essence of the "coarse-filter" conservation strategy, which the United States Nature Conservancy estimates could protect 85–90% of species by conserving samples of natural ecosystems without separate inventory and management of each species (Noss et al. 1995). The conventional "fine filter" of individual-species inventory and protection can address species not dealt with adequately by the coarse filter. Such species of management concern could include rare/endangered and narrow endemic species, large carnivores, and commercially important or culturally significant species. Nevertheless, we must not forsake research at species, population, and genetic levels. We need to know more about the autecology and genecology of selected species at all trophic levels, and we must continue to validate assumptions and test hypotheses about the way biological systems work and how they are affected by human activities and changing environments.

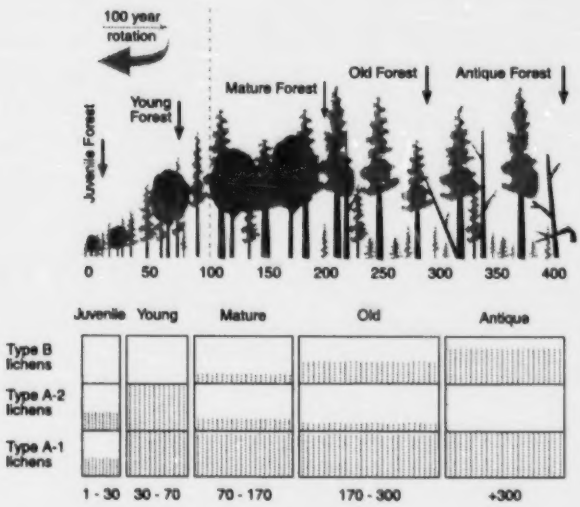

Figure 1. Stand age, development stage, and relative abundance of epiphytic macrolichens in toe-position, western-hemlock-dominated forests of the Kispiox Valley, west-central British Columbia. Type A-1 (terminology of Goward 1994) lichens are widespread generalists that occur in forests of all ages and developmental stages. Type A-2 lichens are also widespread species, but are restricted primarily to younger forests. Type B lichens are specialists restricted primarily to old-growth forests.

ACKNOWLEDGEMENTS

T. Goward and A. MacKinnon helped formulate some of these thoughts.

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Why British Columbia Needs a Provincial Act to Protect Species at Risk and Their Habitats

James N. M. Smith
Centre for Biodiversity Research and Department of Zoology, University of British Columbia
6270 University Boulevard, Vancouver, BC, V6R 3B1, Canada
smith@zoology.ubc.ca

ABSTRACT

Both Canada and British Columbia currently lack specific legislation to protect species at risk. I review the background to this deficiency and comment briefly on the form of upcoming federal endangered species legislation. I propose that a stand-alone British Columbia Endangered Species Act would complement the new federal legislation ideally, and would be the most appropriate way to fulfil British Columbia's obligations under the National Accord for the Protection of Species at Risk. Finally, I consider the form and content of such an act, and whether British Columbia currently has the capability to implement such an act.

Key words: Canada, Endangered Species Act, legislation.

Modern enthusiasm for conservation has deep roots (Norton 1987, Meffe and Carroll 1997), but the subject came of age politically and academically in the 1980s. In that decade, influential academics like Paul R. Ehrlich, Michael E. Soulé, and Edward O. Wilson dedicated their careers to explaining the nature and global extent of conservation problems (e.g., Soulé 1986, Wilson 1988), and mentored an energetic generation of younger scientists. These people firmly established the fact that biological diversity is being lost and some of the causes of the losses (summaries in textbooks by Primack 1993, Hunter 1996, Caughley and Gunn 1996, Meffe and Carroll 1997).

At about the same time, non-government organizations interested in conservation raised public awareness about habitat loss and degradation, and lobbied governments to address these problems. New remote-sensing techniques, particularly LANDSAT images, allowed the extent and severity of global habitat losses to be visualized by all informed citizens (e.g., Skole and Tucker 1993).

Subsequent studies of the patterns of biological extinction (e.g., Pimm et al. 1995, Steadman 1995) have revealed that species and populations are lost when the restricted ranges of endemic species and populations coincide with severe, human-induced modifications of the environment. Although Canada is fortunate in having relatively few endangered organisms compared to tropical and island nations, and even the adjacent United States, we are nevertheless prone to the same processes that generate extinction elsewhere.

By the early 1990s, political leaders in many countries began to recognize publicly that the effects of human population growth and consumption of natural resources by developed nations had become globally unsustainable. This realization led in turn to several international summits, in particular the pivotal Rio Summit in 1992.

THE PROTECTION OF ENDANGERED SPECIES IN CANADA

When Canada played a leading role in promoting the Biodiversity Convention at Rio De Janeiro in 1992, there seemed good reason to hope that a cascade of regulatory and legal safeguards would be rapidly set up for endangered species within Canada. Progress, however, has been slow since the convention was signed in 1993. Indeed, our record of providing legal protection for endangered species has been so poor that we are currently facing threats of international legal action for our failure to protect our species at risk.

A full understanding of why we have failed to create satisfactory protection for Canadian wild species is beyond the scope of this essay, but a brief summary of some key events may be useful. After the failure of the Meech Lake Accord in 1990, successive national governments have moved to devolve responsibility for many key regulatory matters to the provinces. One such matter is the protection of species at risk. A key federal-provincial agreement, the National Accord for the Protection of Species At Risk, was signed by provincial and federal wildlife ministers in October 1996. The next year a federal Endangered Species Act (ESA), Bill C-65, died on the order paper when a federal election was called. A new federal ESA has been promised within the

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

term of the current Liberal government and its content has already been widely discussed at stakeholder meetings, public information sessions, and electronic forums.

THE LIKELY FORM OF A NEW FEDERAL ACT

The new federal ESA is likely to be tabled in the summer of 1999 and to pass into law in the fall or winter of 1999 or 2000. Judging from information provided by Environment Canada (http://www.ec.gc.ca/cws-scf/forum/frame.htm), the new bill will have the following features: 1) it will apply only to federal lands; 2) it will protect the "residences" of listed species and penalize those who kill listed species or willfully damage these residences, but it will not explicitly require the identification and protection of critical habitat for listed species; 3) its formal listing process will resemble the current unofficial one carried out by COSEWIC (Committee on the Status of Endangered Wildlife in Canada), a group of 26 federal and provincial appointees and species-specialist chairs (see Green 2000); and 4) it will contain stewardship initiatives for providing funding to protect listed species on private lands.

The forthcoming federal act is likely to have 2 key limitations: it will not apply to provincial Crown lands, where most of our listed species live; and it will leave the identification and protection of critical habitat, the keys to protection of threatened organisms (Caughley and Gunn 1996, Meffe and Carroll 1997), to be decided case by case at the discretion of the federal government of the day. It is also uncertain if species on the current COSEWIC list will immediately gain legal protection under the new act. There is also concern that new listings may be subject to political interference through CESCC (the Canadian Endangered Species Conservation Council, made up of the federal, provincial and territorial wildlife ministers), which will publish all new listings jointly with COSEWIC.

If such a federal ESA is passed, it will delegate most responsibility for managing endangered species to the governments of the provinces and territories, and to First Nations groups with settled land claim agreements. The National Accord for the Protection of Species At Risk (http://www.ec.gc.ca/press/wild_b_e.htm) clearly expresses the intent of developing parallel and effective provisions for protecting species on federal and provincial/territorial lands. Only in this way, can Canada fulfil its full obligations as a signatory of the Rio Biodiversity Convention. To me, the most effective way to achieve this protective net is to pass provincial ESAs that match or exceed in scope the protection provided by a federal ESA.

THE FORM OF A PROVINCIAL ESA FOR BRITISH COLUMBIA

I shall not comment in detail on the legal issues arising when drafting a new provincial law. It is, however, worth noting that Canadian legal traditions stress discretionary actions by the responsible minister, rather than compulsory actions flowing directly from the written law. The desirable features of a provincial Endangered Species Act are the same as those in a desirable federal act: 1) a scientifically-credible listing process; 2) a series of stringent prohibitions on the deliberate killing or harming of listed species and damaging of their habitats on provincial lands, and perhaps also on other lands; 3) a reasonable list of exemptions to the prohibitions (e.g., forestry operations may reasonably, but inadvertently, remove the nests of a listed species like the coastal subspecies of the northern goshawk [*Accipiter gentilis laingii*]); 4) a process that assesses the habitat needs of listed species and that protects or renews these habitats; 5) other actions to help depleted populations of the listed species to recover; and 6) measures designed to promote stewardship of listed species and their habitats on private and aboriginal lands.

Does British Columbia have the expertise to make such an act work? I think we do. Despite recent losses of senior personnel from the Ministry of Environment, Lands and Parks through forced early retirement, many knowledgeable and dedicated people remain in government service. British Columbia is also fortunate to have many fine naturalists (e.g., Cannings and Cannings 1996) and a strong group of independent wildlife ecologists and conservation biologists, many of whom made presentations at this conference. In particular, recent work at the British Columbia Conservation Data Centre (CDC) in the Ministry of Environment, Lands and Parks, and research work funded by Forest Renewal British Columbia, has done much to establish the status of our populations of wildlife at risk. The files of the CDC and the national COSEWIC lists could provide a firm scientific basis for a provincial list of species requiring protection.

The principal threats to British Columbia's species at risk lie where development pressures coincide with the distribution of rare habitats. These rare habitats lie mostly in the grasslands, riparian areas, and ponderosa pine forests of the southern Interior, and in the Georgia Depression ecoprovince, where human activity and population growth are concentrated. While there are conservation problems in forested habitats, these mostly concern vulnerable but still reasonably abundant species like the marbled murrelet (*Brachyramphus marmoratus*) and the grizzly bear (*Ursus arctos*), and poorly-known species like canopy invertebrates (Winchester 2000). What is lacking currently is not scientific expertise, executive ability in government, or public support, but political will, leadership from top levels in government, and sustained public commitment.

OTHER LEGAL AND POLICY OPTIONS

The current British Columbia government seems unenthusiastic about a provincial ESA (according to statements by the Minister of Environment, Lands and Parks, Cathy McGregor, and the Deputy Minister, Jim Walker, at this conference). According to Walker, British Columbia is contemplating revising the Wildlife Act to meet its commitments under the National Accord. Conservationists in British Columbia can perhaps be forgiven some scepticism over the likely effectiveness of this approach, when the existing powers under the British Columbia Wildlife Act and the British Columbia Forest Practices Code have been so little used to list species or protect their habitats. The province of Alberta is also on record as favouring this minimalist approach (Francis 2000). In my opinion, modifying the Wildlife Act is not only likely to be ineffective, but it constitutes a serious violation of the principles of the National Accord, agreed to by the provinces in 1996. There may be other policy and legal options to protect species at risk in British Columbia (an Environmental Stewardship Act?), but they have yet to be debated widely.

CONCLUSIONS

We live in an information age, when knowledgeable and scientifically aware citizens expect much of their governments. Paradoxically, governments now often respond to these public expectations by taking a risk-averse approach to policy matters and by attempting to conceal policy options from the public, rather than debating their merits openly. While a new provincial ESA to protect species at risk will not, by itself, guarantee the protection of species and habitats, it can nevertheless play several key roles. First, it could be part of a broader package of environmental regulations to deal with land use, invasive exotic species, and industrial impacts on the environment. Second, it could assure the majority of British Columbia citizens, who favour a strong approach to environmental protection, that government is genuinely attentive to their wishes. Third, it could signal to industry groups that potentially costly decisions about habitat protection will be made only when scientific evidence requires considered action, and not at the whim of a special interest group. Finally, a strong bill in British Columbia, which has more of Canada's biological diversity than any other province (Harding and McCullum 1993), would be a timely signal to others that 1 key province is willing to implement the National Accord for the Protection of Species at Risk fully.

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Optimism in the Face of British Columbia's Failing Grade for Species Protection

Kate Smallwood

British Columbia Endangered Species Coalition, c/o West Coast Environmental Law
Suite 1001 - 207 West Hastings Street, Vancouver, BC, V6B 1H7, Canada
ksmallwood@wcel.org

Key words: activism, public participation, stewardship.

It's somewhat ironic, but at a conference that is exploring the ever-increasing number of endangered species and endangered ecosystems in British Columbia, I'm going to tell you that there is room for optimism. Even though we are now experiencing the Earth's sixth period of mass extinction—one of the greatest extinction spasms in geological history (Wilson 1992)—I'm going to explain why there is reason for hope. And in a room filled with government staff and contractors, I'm going to tell you that the primary place to look for salvation isn't inside government, but outside.

THE PUBLIC IS WITH US

The first and most important thing to remember is that the public is with us. Provincial and national polling consistently shows strong public support for legal protection of endangered species and their habitat. A recent province-wide poll found that this support remains constant even if some jobs and economic development could be lost (Viewpoints Research 1998).

Most importantly, however, public opinion research shows that British Columbians place incredible value on natural, beautiful British Columbia. British Columbians define their province by its natural attributes and beautiful environment. When asked to reflect on what images represent British Columbia, residents overwhelmingly list characteristics of the natural world. The exceptional diversity of British Columbia's landscape is particularly valued. British Columbians believe that British Columbia's natural beauty makes it unique in the world. Across the province, people describe British Columbia as "the best place in the world" (BC Wild 1997). These views aren't the shifting sands of public opinion, but deeply held values. Values consistent with the work we do. Values consistent with conservation of biological diversity.

THE TRUTH IS OUT THERE

At last. After 2 years of fighting government claims that current measures to protect species are adequate, the "truth is out there." Thanks to a leaked British Columbia Ministry of Environment document, the truth is there for all to see (British Columbia Ministry of Environment, Lands and Parks 1997). On our Web site in fact (www.wcel.org/esc). The document identifies "significant holes" in British Columbia's legislative and policy measures to protect species at risk and concludes that British Columbia is failing on the job; that British Columbia deserves an "F" for species protection. More recent documentation obtained by the Vancouver Sun under Freedom of Information legislation confirms these findings.

Why is a failing grade reason for optimism? A failing grade for species protection provides an imperative for change. It's downright embarrassing when the province with the most biodiversity in Canada is doing the least to protect it. A failing grade, coupled with widespread public support for species and habitat protection, means that continued government inaction is simply indefensible.

GOD ON OUR SIDE

The increasing involvement of religious groups on environmental issues, and endangered species protection in particular, is truly inspiring. Conservation of biological diversity traverses most if not all religious traditions. Although we are probably most familiar with the Christian images of Noah's Ark and "all creatures great and small," from Bahai to Buddhist to Sikh to Catholic to Anglican and Unitarian, protecting endangered species is an act of faith. Conservation of biodiversity isn't just another social issue on the religious agenda; it is a theological imperative for many people. Protecting endangered species is an integral part of protecting and celebrating God's creation.

Conversely, the opposite is also true. The abuse of nature is a sin.

"It follows that to commit a crime against the natural world is a sin. For humans to cause species to become extinct and to destroy the biological diversity of God's creation; for humans to degrade the integrity of the Earth by causing changes to its climate, by stripping the Earth of its natural forests, or destroying its wetlands...; for humans to contaminate the Earth's waters, its land, its air and its life, with

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb.,1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

poisonous substances—these are sins." These words come, not from an environmental activist, but from the leader of the Greek Orthodox Church at an international Symposium for Sacredness of the Environment (His Holiness 1998).

The power and resonance of religious messages is incredible. It moves the devout and agnostic alike. The potential power of the message is even more remarkable when you realize that religious groups represent the largest grassroots organizations in North America. Previously, religious organizations in Canada have not been active or outspoken on the environment. This is changing, especially in British Columbia.

The interface between religious and environmental groups is particularly interesting. Religious groups are not just another "rent-a-constituency" for the environmental movement. As with the environmental movement, religious groups bring a spectrum of different views and approaches to conservation. In general, the approach is softer and less political, although there are more activist groups in the religious community as well. One of my personal favourites is an American group, the Jesus People Against Pollution. There is no doubt who these folks are and what they want!

Most importantly though, religious groups bring a renewed sense of hope and inspiration to a somewhat jaded and negative environmental community. Religious groups also instill a sense of compassion for people as well as the environment. As British Columbia's traditional resource-based economy shifts to a more diversified modern economy, and workers and rural communities experience rapid transition, this "human face" on environmental protection is very important.

The British Columbia Endangered Species Coalition is actively engaged in outreach to religious groups. I am particularly excited about our involvement with Visions for the Earth. Originally a project of VISION TV (the Canadian religious television network), Visions for the Earth brought people together from the faith, cultural, and environmental communities last Thanksgiving weekend. Together we celebrated the Right to a Safe and Sacred Environment. Paul Gorman from the United States National Religious Partnership for the Environment (NRPE) was the keynote speaker. The NRPE has been very active on a number of environmental issues, including conservation of biological diversity and species protection. The NRPE represents over 150,000 congregations across America.

Visions for the Earth was intended to be a weekend event. Instead, the project has blossomed into an active and expanding coalition of religious, cultural, and environmental groups. We are now working to establish some form of British Columbia Religious Partnership for the Environment. Although the nature and structure of the organization has still to be determined, it is clear that the British Columbia coalition will be a powerful voice for the environment and conservation of biological diversity.

THE VOICE OF THE FUTURE: CHILDREN AND YOUTH

Anyone feeling jaded about conserving biodiversity should spend some time with children and youth. Politicians may not get it, but younger people definitely do. For example:

Ian (Grade 5) says: "Why I feel it is important to protect endangered species. I think it is important because if we don't protect them well there won't be any animals here. If there wouldn't be any animals, this place would be very very very boring." (Peregrine Falcon Recovery Project n.d.)

Allison (Grade 3) says: "Wildlife is great. Imagine: you were sitting down. The sun was bright. The flowers bloomed and you looked at the bright blue sky and saw a peregrine falcon. You watched it fly over your head. A feather came sailing down. Then some black footed ferrets came through the woods playing. The trees were green. Now open your eyes. Now the peregrine falcon is not there. The black footed ferret are hiding. You decide to go out and help nature." (Peregrine Falcon Recovery Project n.d.)

Absolutely. And Allison, I have some copy writing I'd like you to do for us. Instinctively, even with limited knowledge, most kids want to protect and conserve the natural environment. And when they have knowledge, watch out! One of the most moving and articulate pleas for species protection I have ever seen or heard came from a group of children who had discovered tailed frogs in West Vancouver, a remarkable feat in itself. Concerned about the potential impacts of a major housing development on the tailed frog and its habitat, the teenagers attended several West Vancouver council meetings and spoke out for the frog. Not only did the kids delay the development until further studies had been done, they secured additional buffer zones around the creek and educated the Lower Mainland public about the frog and its habitat requirements.

Concern for the environment doesn't diminish with the onset of puberty. Public opinion research shows that youth aged 18–24 are also concerned about our environment and supportive of measures to protect it. The Sierra Youth Coalition has identified endangered species protection as a priority and is actively mobilizing youth on this issue. We are delighted to be working closely with the Sierra Youth Coalition on their campaign.

THE PUBLIC IS WITH US

I'm going to return now to my first point. The primary reason for optimism. The public is with us. I know this for a fact because our British Columbia Endangered Species Slide show is having an amazing effect on people. By simply showing people British Columbia's remarkable biodiversity, explaining what is at risk in British Columbia and why, and why people should care, we can inspire and empower people.

Very easily. With just 4 small presentations in January, we've signed up another 100 supporters. Our slide show will soon be available for loan or purchase at cost, and will be shown by our member groups and interested people across the province. Beautiful, natural British Columbia coming soon to a hall, school, or living room near you.

Conservation of biological diversity will not be achieved overnight. It will not be achieved by provincial or federal endangered species legislation alone, even though these are critical tools and long overdue. It will certainly not be initiated by government. Conservation of biodiversity will only come about when we connect people back to nature. When people understand, they will care. When people care they will act to protect.

This is where you folks come in. The research and inventory work you all do is critical to knowing what's out there, what's at risk and why. But! If the public remain largely ignorant of your work and its importance, our cause will not advance. The conservation of biological diversity will remain a distant goal, constantly thwarted and diverted by tunnel vision from government and competing interests.

Our B.C. Endangered Species Slide Show is our small contribution to connect British Columbians to this remarkable place we call home. To touch their sense of pride, to empower them to act.

I'm going to conclude this ode to optimism by celebrating supernatural British Columbia with you—what many British Columbians regard as "the best place in the world." And with this brief extract from the beginning and end of our slide show, I'm going to ask you to act as well—to join the chorus. And remind you that you will not be alone.

BRITISH COLUMBIA'S REMARKABLE BIOLOGICAL DIVERSITY

[The following text is a transcript of the author's commentary, which accompanied the images of the British Columbia Endangered Species Slide Show.]

So in closing, let's celebrate a very special place. A place larger than any European country, except Russia, and any U.S. state except Alaska. A place with biodiversity of national and international significance. And the best part about it? This place is our place. This place is British Columbia.

The amazing news is that British Columbia has more living things than any other province or territory in Canada. WOW! Our province is home to over 70% of all the bird and mammal species in Canada, and about half of these species, like our very own Vancouver Island marmot, are exclusive to the province. We have more large mammal species than any other province or state in North America, and the most diverse, intact predator-prey systems on the continent. British Columbia's bag of riches doesn't stop there.

We have 49% of Canada's amphibian species. This is the tailed frog. The coastal population of this species is classified as vulnerable and the Kootenay population is classified as endangered.

And 41% of Canada's terrestrial reptile species. This is the vulnerable yellow-bellied racer.

Furthermore, with over 3,150 species of plants, we also have the richest flora in Canada. WOW AGAIN!

As if all this wasn't enough, for many other species, British Columbia is also home to most of the world's population. For example: 60% of the world's mountain goats are found here, along with 30% of all bald eagles, and about 25% of all grizzly bears.

British Columbia also has some of the largest intact, temperate rain forest in the world: areas that have been classified by World Wildlife Fund as "globally outstanding" ecoregions.

Unfortunately, this incredible biodiversity, like this threatened white-headed woodpecker, is at risk. The latest figures from the B.C. Ministry of Environment show there are 345 endangered and threatened species in our province. Another 460 species are vulnerable. In total, there are 805 species at risk in British Columbia. These species are like fire alarms, warning us that things are wrong with our natural environment.

So is there any good news from this catalogue of British Columbia species at risk? The answer is a RESOUNDING YES! Over half of British Columbia's 805 species at risk, like grizzly bears, are only vulnerable. This means we have the opportunity—if we act now—to reverse the path towards endangerment and extinction.

What you can do to help: The most important thing to remember is that YOU, YES YOU, can make a difference. The future of our planet and our endangered species lies in your hands The last federal endangered species bill didn't come about because the federal government wanted it. Federal endangered species legislation was put on the table because over 80,000 Canadians wrote letters, sent cards, or signed petitions demanding it. Opponents of endangered species legislation are relying on your complacency. They're hoping that you'll just lie around. That you won't care if all we get is weak federal legislation, or in British Columbia's case, no legislation at all.

Well this slide show is designed to change that. To get under your skin a bit. To get you moving on this issue. To encourage you to speak out. To join the growing chorus for decent endangered species protection both federally and right here in British Columbia. Your voices do count. You are not alone in the wilderness on this issue. One letter from you to a politician is held to represent the views of at least 400 other people. And when we act collectively, in a targeted fashion, our voices are even louder and more effective.

Joining the chorus is really easy. If you are prepared to make several phone calls and write the occasional letter over

the next 6 to 12 months, we can get you organized. You can join the chorus by signing up for our endangered species e-mail listserve or our member newsletter. It's that simple.

I'm going to close this slide show with the image of British Columbia's Spirit Bear and words from a beautiful Pablo Neruda poem (Neruda 1972). The poem helps remind us all that from small beginnings, great feats can be achieved with the weight of the people behind you.

and I walk steadily forward
because I have a thousand eyes,
I strike with the weight of a rock,
because I have a thousand hands
and my voice is heard on the shores
of all the lands
because it is the voice of all
those who did not speak
of those who did not sing ...

Please join us and add your voices to the chorus. Thank you.

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Closing Remarks for the 1999 British Columbia Conference on Biology and Management of Species and Habitats at Risk

Don Eastman
565 Beaver Lake Rd., RR#3, Victoria, BC, V8X 3X1, Canada
deastman@uvic.ca

ABSTRACT

The closing remarks for the conference address 3 questions: 1) How far have we come?; 2) Where are we now?; and 3) Where to from here? Using the proceedings of a 1980 conference on the same theme as a reference point, it is clear that species and habitats at risk now command much more interest than before; that the level of knowledge about these topics has increased substantially; and that the conservation of habitats and species has improved. Notwithstanding these significant accomplishments, much remains to be done, because the situation for species and habitats continues to deteriorate. Existing legislation needs to be more effectively applied and new legislation on endangered species is needed. Conservation biologists and managers need to re-assess how they balance the need for additional information with the need for action. Greater effort is needed to educate all sectors of the public on conservation issues. Significant knowledge gaps remain to be filled, and a more secure funding base is a prerequisite for this need to be met. Improved communication and integration is needed among the scientific disciplines that make up the conservation field, as well as between the various activities, such as research, management and education. Finally, a strong need exists for greater cooperation among the many and diverse organizations that span the conservation spectrum.

Key words: education, integration, legislation, research, review, risk.

I appreciate the honour and the opportunity to be the rapporteur for our meeting, but I am not so sure about the required effort...especially for a retired person such as myself! In any case, I began this exercise by asking, what is a rapporteur? According to my Canadian Oxford Dictionary (Barber 1998), rapporteurs summarize the work of a group for a higher authority. Now, I have difficulty with higher authorities unless I respect them, so I have arbitrarily selected my authorities as the conference co-chairs, Lord Fraser and Lord Larsen.

My remarks address the following three questions: 1) How far have we come?; 2) Where are we now?; and 3) Where to from here? But before I begin, some introductory remarks are in order. This has been an exceptionally fine conference and this has not happened by chance! If I did nothing else this afternoon, I would want to extend thanks:

First, to the 6-member organizing committee, which spent 2.5 years planning this conference—it was a long gestation—and I would also include in this group those people at the University College of the Cariboo who looked after us so well during these past 3 days.

Second, to the volunteers, for without their help this conference could not have happened.

Third, to the more than 70 sponsors and contributors that provided the funding and other essential, tangible support that enabled a conference of this size to become a reality.

Fourth, to the speakers, who shared their results, ideas, and approaches for our benefit.

Fifth, to the audience, for without the audience there would be no conference.

It was only through the collective effort of all of you that this meeting has been so successful and so rewarding.

CONFERENCE OVERVIEW

This meeting is one of those gatherings that will be talked about in the years to come. If we could step into a time machine and fast-forward a few years, I am sure we would hear comments such as: "Were you at the Kamloops meeting about endangered species and habitats?"; "Remember Dave Green's frog saying thank you"?; "Remember Geoff Scudder's keynote address? It was excellent!"; "Remember Neville Winchester's broadside about reports with no science!"; "Remember Pat Slaney informing us that we live with impacts for a hundred years?"; "Remember Terry McIntosh disclosing that he was nervous and excited?" This meeting will enter our collective memory as a significant milestone.

L. M. Darling, editor. 2000. Proceedings of a Conference on the Biology and Management of Species and Habitats at Risk, Kamloops, B.C., 15 - 19 Feb., 1999. Volume Two. B.C. Ministry of Environment, Lands and Parks, Victoria, B.C. and University College of the Cariboo, Kamloops, B.C. 520pp.

Just what did this conference have to offer? It offered a concentrated learning opportunity on a great many topics—in the lecture rooms and in the corridors, both planned and fortuitous. It offered an opportunity to meet new people, some of whom were only names in journals, and to re-connect with others. Isn't it amazing that even with the remarkable technologies of contemporary communications, we still need that face-to-face contact! It offered energy and enthusiasm—a chance to re-charge those depleted batteries and head back out to the trenches. It offered an opportunity to step outside our daily routines and re-confirm our place and importance in the ongoing conservation of species and habitats at risk. Above all, this meeting offered renewal.

HOW FAR HAVE WE COME?

To answer this question, we need a benchmark, a point of reference from which to make comparisons. Fortunately, we have a benchmark: in March 1980, at Simon Fraser University, there was a symposium on endangered species (Stace-Smith et al. 1980). We can make some interesting comparisons between then and now, a generation later.

First, what about the level of interest? The 1980 symposium attracted 250 people in the most heavily populated area of the province. The 1999 conference has an official registration of 450, and we had more than 600 hundred people in the public session. Another perspective on the level of interest is possible if we compare the number and types of sponsors. In 1980, there were 3 sponsors (Federation of BC Naturalists, Douglas College and the British Columbia Fish and Wildlife Branch), but in 1999 there were more than 70 sponsors, from government, industry, and a variety of not-for-profit conservation organizations. To be sure, there has been a tremendous growth in interest in species and habitats at risk.

Second, what about the scope of topics? The 1980 meeting dealt mostly with vertebrates. While the "vertebrate bias" is still with us, our scope has expanded in depth and breadth. This week we had sessions on freshwater mollusks, forest-canopy arthropods, dragonflies, lichens, and macrofungi. What a feast!

Third, what about the gain in knowledge? At the 1980 symposium, 29 papers were presented. The 1999 symposium had 3 plenary sessions containing 15 presentations, plus 4 to 5 concurrent sessions each day for a total of 145 papers, plus 40 posters and 3 public meetings! From this comparison alone, it is clear that we have made significant gains in knowledge.

Probably the most important comparison is our progress on conservation issues. We can look at this from habitat and species perspectives. From a habitat perspective, we now speak of ecosystem-based management, a term that wasn't even in our lexicon in 1980, and this belies a broadened awareness of the interconnectedness of the biophysical world. We have the Protected Areas Strategy, with a governmental commitment to protect 12 percent of the province's landscape, representing all major terrestrial ecological zones, and we are even beginning to think about marine ecosystems. We have the Forest Practices Code, touted as the best in the world, which deals with the conservation of biological diversity—at least on paper. We have the Conservation Data Centre, which provides objective and authoritative assessments of the status of ecological communities. Again, I believe we can say there has been progress.

How well have we done in conserving endangered species? From the 21 recommendations of the 1980 symposium I have selected 7 that dealt with British Columbia:

1. The 1980 symposium recommended that we learn more about lesser known fauna and, as this meeting has shown, many other taxa have been studied since then. However, there still is a strong vertebrate bias!
2. The 1980 symposium recommended that we include peripheral species and subspecies in assessing status, and this is being done.
3. The 1980 symposium recommended that a board be established to determine which species should be designated. We have no board per se, although nationally we have the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) that designates the status of species for Canada. Provincially, we have the Conservation Data Centre that uses a standard, widely accepted protocol to determine the status of vascular plants, vertebrates, selected invertebrates, and ecological communities.
4. The 1980 symposium recommended that the Ministry of Forests be responsible for plants and insects at risk. At this time there is no legislation to this effect, but the Forest Practices Code includes the Ministry of Forests, in partnership with the Ministry of Environment, Lands and Parks, in the management of species and habitats at risk, including plants and invertebrates.
5. The 1980 symposium recommended that more attention be given to the following endangered species: burrowing owl, peregrine falcon, sandhill crane, spotted owl, white pelican, and sea otter. Progress has been made on all of these species, but only 1, the sea otter, has been de-listed, and the statuses of the burrowing owl and sandhill crane have deteriorated. (The Vancouver Island marmot, the province's most endangered species, was not even mentioned!)
6. The 1980 symposium recommended that we address the problem of exotic species. Very limited progress has been made on this front. British Columbia still does not have adequate provisions for dealing with exotics, and we continue to cope inadequately with this very serious problem.
7. The 1980 symposium recommended that legislation be enacted that establishes the right of all species to exist for

their own sake. This has not been done. We have no endangered species legislation at either the provincial or federal level, and no new species have been designated as endangered under the Wildlife Act since 1980.

As I look over this list, I am reminded of a sobering comment offered by Ian McTaggart Cowan in his introduction to the 1980 symposium: "All the evidence emphasizes that North Americans, with their concept of local autonomy in resource management, have established the worst record of any continent in the destruction of native animals." (Cowan 1980:18). So, overall, how far have we come in protecting species at risk?

WHERE ARE WE NOW?

At this time, we have made significant gains, both in our knowledge base and in our capacity and expertise. Think about the many individuals with education and experience related to endangered species and habitats. (Note, however, that a lot of this is quite recent, stemming from funding by Forest Renewal British Columbia.) There have been significant advances in technology—GPS, GIS, satellites, and powerful computers—that help in advancing our understanding of species and habitats at risk. As well, we have made valuable conceptual advances. Consider the application and scope afforded by such concepts as viable population, metapopulation, connectivity, and biodiversity. Think about the emergence of fields such as conservation biology and landscape ecology—these areas of endeavour were not even invented in 1980 (although many people were working in areas now included by these terms).

But, there remain some sobering points to bear in mind. For example, the number of species and habitats at risk continues to increase. This increase results partly from a greater awareness (we know more now), but, more significantly, from increasing pressures on species and their habitats. We are still dealing with issues that were defined and active more than 20 years ago. For example, the South Okanagan is still dealing with the loss of habitat from human settlement, although major gains have been made. Species such as the burrowing owl, Vancouver Island marmot, and spotted owl are still not secure, demonstrating how difficult it is to recover species at the edge. In other words, the fundamental and proximate causes of the crisis are still with us—locally, provincially, nationally, and globally. It is not the time to rest on our laurels!

WHERE TO FROM HERE?

To speculate on this question is overwhelming and possibly even presumptuous. I approached this question by drawing upon discussions with others, by eavesdropping on conversations, and by reflecting on the presentations we have heard over the past 3 days. From all of these sources, I came up with 6 main ideas. You can credit me for those ideas you think are good... and blame the others for those you think are worthless!

First, we need to improve our legislative framework. We need to apply existing legislation more effectively than we do now. The Wildlife Act of British Columbia has provisions for designating endangered species, but none have been listed since the initial 4 in 1980, and we still do not list plants and invertebrates. There are other examples that indicate the significance of political will to fully implement the powers of existing legislation. Another point is that we need to amend and improve laws and regulations already in place. A good example of this relates to the importation of plants and animals into the province. We need a change in regulations that specifies a "clean" list, that is, a list that designates those species that are allowed into British Columbia—all others are prohibited. Finally, we need new legislation. Notwithstanding the provisions of existing laws, I believe that we need an endangered species act; we still need improved protection of habitat; and we would benefit from legislation that makes biologists licensed professionals, similar to foresters, engineers, and agrologists.

Second, we need to deal with risk differently. Many of us were trained in the scientific method, to set up testable hypotheses and conduct critical experiments before we came to conclusions. While this approach is very powerful, it can make us risk-averse. In the field of conservation, where we deal constantly with uncertainty and lack of knowledge, a tension exists between the need to do something and the risk of being wrong. However, when a species' existence is at stake, it may be better to do something that turns out to be unnecessary, rather than do nothing and watch a species go extinct—we must deal with this asymmetry of outcomes.

Third, educate, educate, and educate! As biologists largely focussed on research and management, we tend to ignore or play down our role as educators. More of us need to step out of our traditional roles and communicate effectively—in the field, in the schools, and in the boardrooms of decision-makers. Now, for a variety of reasons, not all of us can assume these roles and, with these individuals, the least we can do is improve our links to environmental educators and interpreters who have the communication skills and abilities and an ever-growing audience. The need is greater than ever and does not go away.

Fourth, we need to continue expanding our existing knowledge base. Many gaps still exist across the broad conservation spectrum—alien species, understudied taxa, the small population paradigm, and the design of protected area systems—to mention just a few topics. We need to do this so that we can continue to bring the best available science to bear on conservation issues. Credibility is the key, for even though solutions to conservation issues must consider social

values and economics, the scientific basis is fundamental.

Fifth, we need better ways to secure adequate resources for conservation. The potential benefits of adequate funding are demonstrated by the impact of funding from Forest Renewal British Columbia on ecological studies in this province—how many papers at this conference acknowledge support from this funding source? Currently, we have tremendous capacity in terms of expertise, but without predictable and sufficient funding we will continue to be opportunistic and short-term in our approach to research. This is not the paradigm we need to answer long-term conservation questions. Funding sources, such as the provincial Habitat Conservation Trust Fund, have made a real difference in wildlife and fisheries management. Perhaps we need to expand upon this fund, or develop a parallel one aimed specifically at endangered species and habitats.

Finally, we need to integrate more effectively across all the fields and subjects of conservation. A hallmark of our chronic issues is fragmentation among bureaucracies, and among interest groups. We need to think of conservation as an arch—each stone in the arch is necessary, but none is sufficient to make an arch. All parts are essential. We can apply this image in several important contexts. One application relates to scientific disciplines, where we need greater integration amoung fisheries, wildlife biology, ethnobiology, palynology, forestry, and agrology, because our problems are largely ecological and multi-faceted. Another application is to think of the arch in terms of components of a conservation strategy. We need contributions from research, education, protected areas, management, legislation, and other fields. Research by itself is insufficient, for it only explains; legislation alone will not solve the problems, because some problems are not legal; protected areas are key, but they will never occupy more than small proportions of land and water. Because conservation issues are complex, their solutions demand attention from more than 1 domain to be successful. Finally, we can think of the arch in terms of organizations involved in conservation. In British Columbia we have hundreds of groups that care about protecting and conserving the natural world. While it is important to have "homes" for the many viewpoints and interests extant in our society, it is just as important to realize that it is the collective sum of efforts that will provide the impetus for meaningful change. But, what is the keystone for all of these 3 contexts? I suggest that the keystone is our common commitment to the conservation goal.

I would like to close with a recommendation that we need a conservation conspiracy (thanks to Biruté Galdikas for this idea). We need to adopt the long view; we need to hang in for the long term. As Pat Slaney remarked, we deal with long-lived issues, and our failures (and our successes) have long-term consequences. We need to weather the test of adversity. We need to do good, challenge the status quo, and, especially, support each other. For all our apparent differences, we ultimately are working towards the same goal.

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